From salt to C; carbon sequestration through ecological restoration at the Dry Creek Salt Field

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Goyder Institute for Water Research
Technical Report Series No. 19/28

www.goyderinstitute.org

Goyder Institute for Water Research Technical Report Series ISSN: 1839-2725

The Goyder Institute for Water Research is a partnership between the South Australian Government through the Department for Environment and Water, CSIRO, Flinders University, the University of Adelaide, the University of South Australia and the International Centre of Excellence in Water Resource Management. The Institute enhances the South Australian Government's capacity to develop and deliver science-based policy solutions in water management. It brings together the best scientists and researchers across Australia to provide expert and independent scientific advice to inform good government water policy and identify future threats and opportunities to water security.

This project was carried out in collaboration with the Australian National University.

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Citation

Dittmann, S., Mosley, L., Beaumont, K., Clarke, B., Bestland, E., Guan, H., Sandhu, H., Clanahan, M., Baring, R., Quinn, J., Seaman, R., Sutton, P., Min Thomson, S., Costanza, R., Shepherd, G., Whalen, M., Stangoulis, J., Marschner, P., Townsend, M. (2019) *From salt to C; carbon sequestration through ecological restoration at the Dry Creek Salt Field.* Goyder Institute for Water Research Technical Report Series No. 19/28.

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Executive summary

The storage and sequestration of carbon by saltmarsh, mangrove, and seagrass beds is per unit area higher than of any other vegetated habitat and is referred to as blue carbon (Herr et al. 2011; Howard et al. 2017; Nellemann et al. 2009). Blue carbon has been identified as a key carbon offset opportunity for South Australia. As part of the Climate Action Research Impact Area of the Goyder Institute for Water Research, this project investigated whether tidal reconnection and restoration of the Dry Creek salt fields can be a pathway towards realising blue carbon opportunities for South Australia.

The cessation of salt production at the Dry Creek salt field provided an ideal study case for reconnection, realised through the installation of a tidal gate in a trial pond. This reconnection was new, additional and met the offsets integrity standards for a project activity under the Emissions Reduction Fund (ERF). It can thus provide a proof of concept and pathway for future projects introducing tidal flow.

In an interdisciplinary approach, the project investigated the blue carbon potential and co-benefits of coastal wetlands. Over 1.5 years post reconnection, changes in several carbon pools were investigated inside the trial pond and in reference areas. Assessments were made for several strata, defined by elevation and predicted vegetation classes. The processes of revegetation were experimentally studied. Findings from all field investigations were combined to calculate current carbon stocks and sequestration rates, and project potential benefits forward in time and space through upscaling for several scenarios.

The investigations on carbon dynamics showed a net gain of soil organic carbon stock following tidal reconnection, which could be partly attributed to influx of seagrass wrack. Methane gas fluxes were negligible and could be excluded from further carbon pool assessments. Sediment accumulation rates were highly variable across the strata and between the trial pond and reference areas. The carbon fraction for saltmarsh and above and below-ground vegetation biomass was determined. While the greatest carbon capture was in mangrove, saltmarsh was an important contributor to carbon sequestration.

Saltmarsh vegetation rapidly colonised the pond following tidal reconnection, dominated by pioneer species *Suaeda australis* and *Sarcocornia quinqueflora* which grew quickly to mature stages inside the trial pond. Experiments revealed the dependence of revegetation on nearby seed supply, and seasonal variation in seed dispersal. A net gain in carbon stock from the establishing saltmarsh inside the pond was estimated.

Based on the measurements from this project, and informed by a further study on a nearby chronosequence, the net project benefits were estimated at the scale of the trial pond, and for scenarios that hypothetically reconnected additional ponds of the salt field. It was estimated that in 30 years, the total carbon stock could potentially reach about 463 000 t CO_2e for an area of 1210 hectares of reconnected low-lying ponds, and over 652 000 t CO_2e for an area of 1963 hectares with additional ponds reconnected at supra-tidal elevation. The respective net gains for change in carbon stocks (soil and biomass) under project scenarios above the business-as-usual baseline, could be over 218 000 t CO_2e for 1210 hectares of reconnected ponds, and over 250 000 t CO_2e for the 1963 hectares area within 30 years.

Carbon sequestration from tidal reconnection is one of the benefits that can be derived from restoration of salt ponds, with benefits increasing the larger the restored area. The habitat provided by restored saltmarsh and mangrove will enhance ecosystem services with relevance for human well-being. A social survey revealed strong cultural ecosystem values associated with the coastal wetland region north of Adelaide. Restoring the salt field to tidal wetlands can thus yield multiple benefits.

In the absence of a suitable method for blue carbon under the Emissions Reduction Fund (ERF), an attempt was made to register the trial pond under the *Human-Induced Regeneration of a Permanent Even-Aged Native Forest* (HIR) ERF method. The Clean Energy Regulator (CER) did not approve the application as FullCAM

(Full Carbon Accounting Model) is not able to model mangroves. While the trial pond project could not be successfully registered as an eligible carbon offsets project with the CER, it has become a pilot project for the introduction of tidal flow as an activity under a potential new blue carbon ERF method in development.

The project has delivered on the intended outcomes for addressing knowledge gaps:

Understanding the value of coastal environments as carbon sinks – we gained data and knowledge on carbon stocks and sequestration rates for coastal wetland soils, mangrove and saltmarsh, and identified processes which enable restoration of coastal wetlands through tidal reconnection. The greenhouse gas (GHG) flux measurements have shown that methane emissions are negligible from the highly saline soils in tidal wetlands.

Spatial extent and condition of coastal environments and their potential levels of carbon emissions and sequestration (for carbon accounting) – data for soil and biomass carbon pools obtained from reference areas for specific elevation strata are valuable for upscaling from study sites to larger areas.

The potential for carbon offsets from coastal environments – the tidal reconnection trial has successfully demonstrated that opening a salt pond to tidal flow can lead to restoration of coastal wetland without adverse effects. Data showed that even after just 1.5 years, the reconnected pond was naturally revegetated by saltmarsh, had increased soil organic carbon content, and GHG emissions were negligible. Once a methodology for blue carbon is available through the ERF, or a solution found for the potential double-counting issue when using international carbon offsetting methods, carbon offset credits can be gained. The offset potential increases the larger the reconnected area.

Development of tools and information to support assessment of the optimum mix of carbon offsets achievable – data from the project will become part of databases for use in national inventory reporting and for improving default values for blue carbon at a regionally specific level.

The project also had impact for policy development:

Support development of a blue carbon strategy for South Australia – this project informed the practicability for a blue carbon strategy for South Australia, specifically the suitability of tidal reconnection as a blue carbon project activity and the application of carbon accounting methods. The project has gained relevance beyond South Australia by informing progress for blue carbon method development under the ERF.

New science that supports the identification of carbon offset opportunities – the project constitutes the first study to investigate effects of tidal reconnection of a salt pond for carbon stocks. The study has shown measurable increases in soil and biomass carbon stocks, which can lead to carbon offset opportunities. The modelling has further revealed that within 30 years, offset opportunities can arise and be substantial when larger areas of the salt field are reconnected.

Quantification of the carbon sequestration potential of coastal wetlands – carbon sequestration rates were determined for mangrove and several elevation strata of saltmarsh. As sediment accumulation rates were highly variable, data are mostly presented as carbon stocks. Quantification of sequestration requires a greater effort to determine the necessary sediment accumulation rates across blue carbon ecosystems.

Identification of the extent, condition and carbon budget of coastal environments to inform national and state carbon accounting – data from the project can be used to inform state and national inventory reporting.

Acknowledgments

This project was possible through funding of the Goyder Institute for Water Research, and we are grateful for the support received. Discussions with the Goyder Institute for Water Research and Joint Project Advisory Committee were also beneficial for advancing the project.

The project location was part of the Dry Creek Salt field, and we are very thankful to Buckland Dry Creek for providing access to the site for carrying out the investigations.

Sampling for the saltmarsh and mangrove vegetation components was carried out under permit number E26725-1.

The saltmarsh vegetation transects were carried out by Doug Fotheringham, with further assistance from Guy Williams, Shari Detmar and Alison Turnbull.

Field and lab work would not have been possible without helping hands, and we greatly appreciate the precious time given by volunteers. Josh Arthur, Alex Blackall, Nick Board, James Christie, Woody Drummond, Sophie Gilbey, Lucy Gilchrist, Orlando Lam-Gordillo, Mathew McCaskil, Laura Schroder, Amy Slender, Bodhi Thomas, Taylor Vogt, Nick Wilkins, Caitlin Wilkinson and Grace Yool volunteered 564 hours of their time to field, glasshouse and laboratory work. Atun Zawadski (ANSTO) helped with the lead-210 dating determinations.

The final report benefitted from the review by Steve Crooks, Louisa Perrin and Jeff Kelleway, and we thank them for their constructive comments.

1 Introduction

1.1 General aim and objectives of the project

To reduce concentrations of greenhouse gases (GHG) in the atmosphere, countries like Australia that are signatory to the Paris Agreement of the United Nations Framework Convention on Climate Change (UNFCCC) have pledged to undertake efforts to mitigate global warming. The Nationally Determined Contributions set by each country can be achieved by emissions reduction and/or carbon offsetting through market mechanisms. The UNFCCC, and other international conventions, have recognised the role of oceans and coastal wetlands as sinks for GHG. The storage and sequestration of carbon by saltmarsh, mangrove, and seagrass beds is per unit area higher than of any other vegetated habitat, and referred to as blue carbon (Herr et al. 2011; Howard et al. 2017; Nellemann et al. 2009) (Figure 1). For South Australia, blue carbon has been identified as a key carbon offset opportunity. As part of the Climate Action Research Impact Area of the Goyder Institute for Water Research, this project investigated whether tidal reconnection and restoration of the Dry Creek salt fields can be a pathway towards realising blue carbon opportunities for South Australia. The South Australian Government is developing a blue carbon strategy which will include pathways for achieving carbon credits for blue carbon projects.

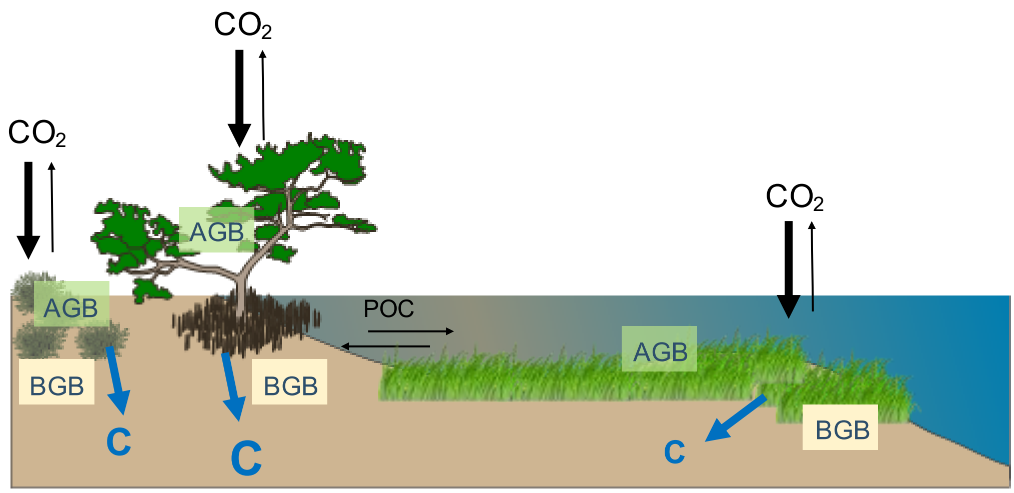

Figure 1. Conceptual diagram of blue carbon sequestration in coastal ecosystems. ABG represents above-ground biomass and BGB represents below-ground biomass.

Tidal reconnection has been identified as one of the most lucrative blue carbon activities for projects under the ERF in Australia (Kelleway et al. 2017a). In South Australia, the cessation of salt production at the Dry Creek salt field provided an ideal study case for reconnection, realised through the installation of a tidal gate in a trial pond (see section 2.1). This reconnection was a new activity, additional and met the offset integrity standard criteria (see section 5). The business-as-usual case which forms the baseline was the continued operation of salt ponds as a salt field.

The project had the ambition to provide a feasibility assessment for blue carbon and further co-benefits from salt field restoration, and show a market ready pathway for carbon accounting for the generation of carbon credits, whether under the Verified Carbon Standard (VCS) or ERF. This proof of concept is presented in an accompanying report (Dittmann et al. 2019). We followed the Verified Carbon Standard (VCM0033 Tidal Wetland and Seagrass Restoration) approach for carbon offset accounting, as documented in the proof of concept report (Dittmann et al. 2019), which builds on research findings presented here, and additional studies. While carbon credits from voluntary markets cannot be used in Australia due to double counting issues, our assessment provides a proof of concept and pathway for future projects under an adapted VCS method for inclusion into the ERF. As an alternative method in the absence of an ERF method for blue carbon, the applicability of using the Human-Induced Regeneration method under the ERF for mangroves forest was briefly evaluated (see Chapter 5).

1.2 Research approach

The approach taken to investigate blue carbon potential and co-benefits from tidal reconnection was interdisciplinary, including participants with expertise in biogeochemistry, botany, coastal wetland ecology, carbon accounting, ecological economics, social sciences, geospatial modelling, and environmental management. The project team carried out field investigations and experiments, comprehensive analyses, workshops, a social survey and modelling. Table 1 provides an overview on the five task groups and their specific objectives and research tasks, which form the structure of this synthesis report.

Table 1: Overview of the project tasks and their specific objectives.

TASK	TASK OBJECTIVES	TASK LIST
1 - Carbon dynamics & sequestration	Sequestration and abatement of carbon through salt field restoration	Carbon stock measurements in sediment and biomass Sediment accumulation rates GHG emission measurements
2 - Revegetation experiments	Feasibility assessment of revegetation and effects on carbon sequestration	Transplantation and seedling experiments Growth and carbon stock measurements
3 - Carbon accounting and offset registration	Carbon offset calculation and registration under the ERF	Evaluation of carbon accounting methods for the Australian context Roadmap for registration Carbon accounting and audit of project data Registration with CER
4 - Co-benefit analysis and up-scaling	Assessment of co-benefits and carbon offset for larger salt field area	Refinement of geospatial database Database for local/regional ecosystem service values and co-benefits Co-benefit analysis Upscaling of carbon offsets and synthesis
5 - Translation of outcomes, pathway to market	Proof of concept	

A conceptual model was developed to integrate the project tasks and illustrate how they will inform the pathway of restoration and blue carbon outcomes. Tidal reconnection was aligned with the concept of 'Windows of Opportunity' (Balke et al. 2014), whereby changes to the physical forcing after tidal reconnection provide a disturbance-free context suitable for vegetation re-establishment, which is leading to carbon stock increases and sequestration and further co-benefits (Figure 2). Parts of the physical forcing and the carbon assessments were realised by tasks 1, 2 and 3, and the biological forcing through the revegetation studies in task 2. Task 4 evaluated co-benefits and further ecosystem services, and extrapolated the research outcomes to a larger scale using several scenarios. Task 5 ensured the communication of findings from this project to decision makers.

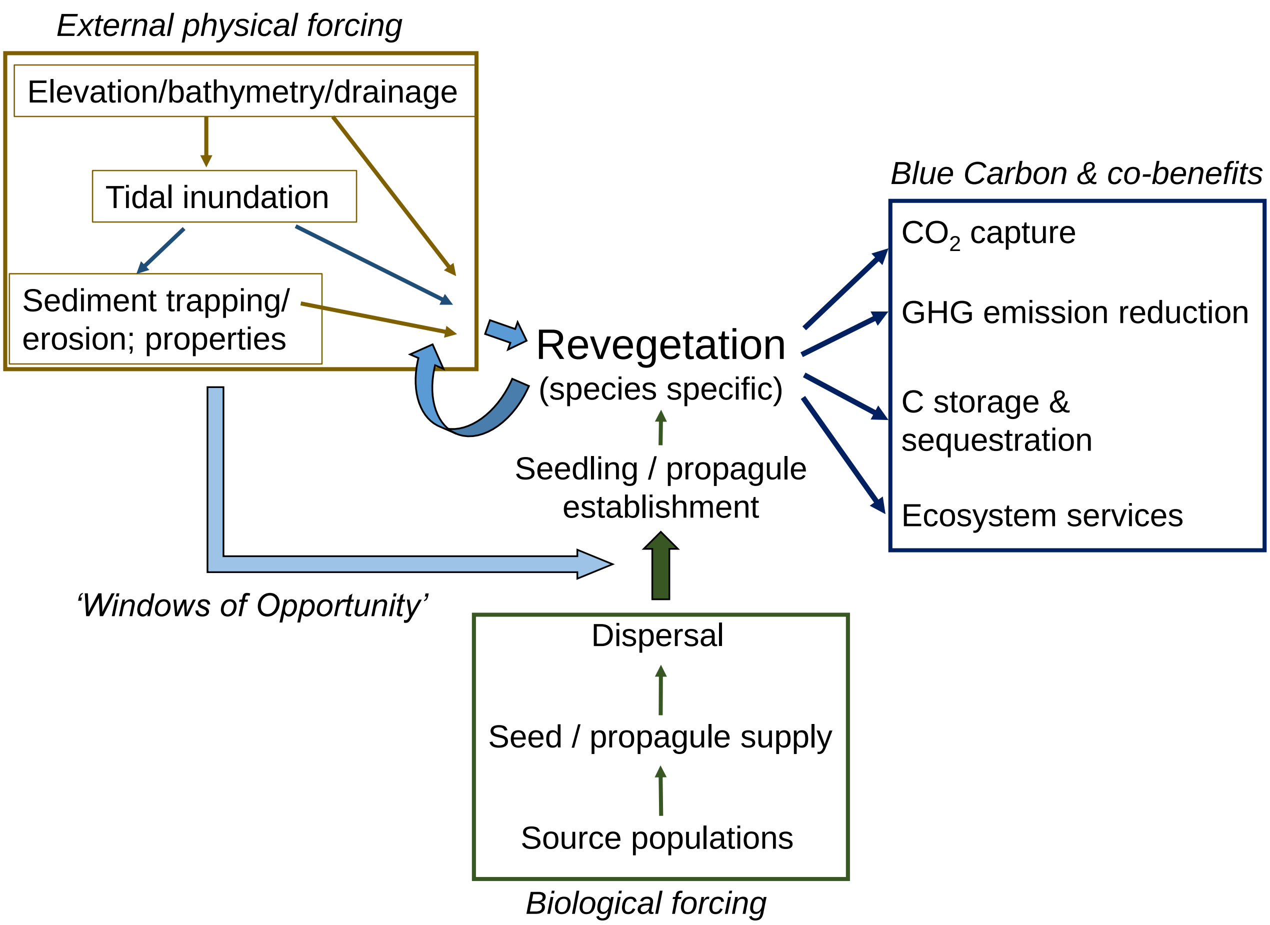

Figure 2. Conceptual diagram of processes enabling blue carbon benefits through tidal reconnection of a salt pond.

1.3 Significance of the project

The relevance of this project for informing climate change policies increased during its duration with the development of a blue carbon strategy for South Australia. A technical review of blue carbon opportunities in Australia's ERF identified tidal reconnection as a top priority activity (Kelleway et al. 2017a). The 'Salt to C' project has thus gained state and nation-wide interest as the first pilot study which can provide a proof of concept for blue carbon benefits from tidal reconnection and salt field restoration. The knowledge gained from the project will inform decision making on coastal restoration in South Australia and nationally, which can enhance ecosystem functions and services, adaptation to climate change, mitigation of GHG emissions and the development of blue carbon methods under the ERF.

2 Tidal reconnection trial

2.1 Reconnection trial - context and realisation

Salts (mainly halite) have been produced at the Dry Creek salt field (4000 ha) north of Adelaide since 1940 (Bell 2014) (Figure 3a). The salt was produced by evaporating seawater pumped into a series of concentrating ponds where water evaporation induced calcium carbonate ($CaCO_3$), gypsum ($CaSO_4$) and then common salt (halite, NaCl) precipitation. One major issue noted during closure of the Dry Creek operations is that hypersaline and sulfide-rich sediments have also built up over large areas which poses a potential environmental hazard (e.g. if resuspended) and a barrier to recolonisation of native vegetation and marine invertebrates (e.g. hyper-salinity and sulfide toxicity). The commercial salt production operation ceased in 2013.

One option to remediate the salt field is to undertake tidal cycling to reduce the environmental hazard. This can be achieved by installing a tidal gate between a pond and tidal creek to allow for carefully managed tidal flushing. The potential benefits of tidal cycling not only include reducing the hypersaline and monosulfidic material hazard (Fitzpatrick et al. 2015; Mosley et al. 2015), but also the regeneration of tidal creeks and saltmarsh/mangrove to provide habitat and food resources (i.e. invertebrates) for birds and fish. The restoration process needs to ensure that impacts on the surrounding environment are minimised as the site is adjacent to important aquatic ecosystems and marine reserves.

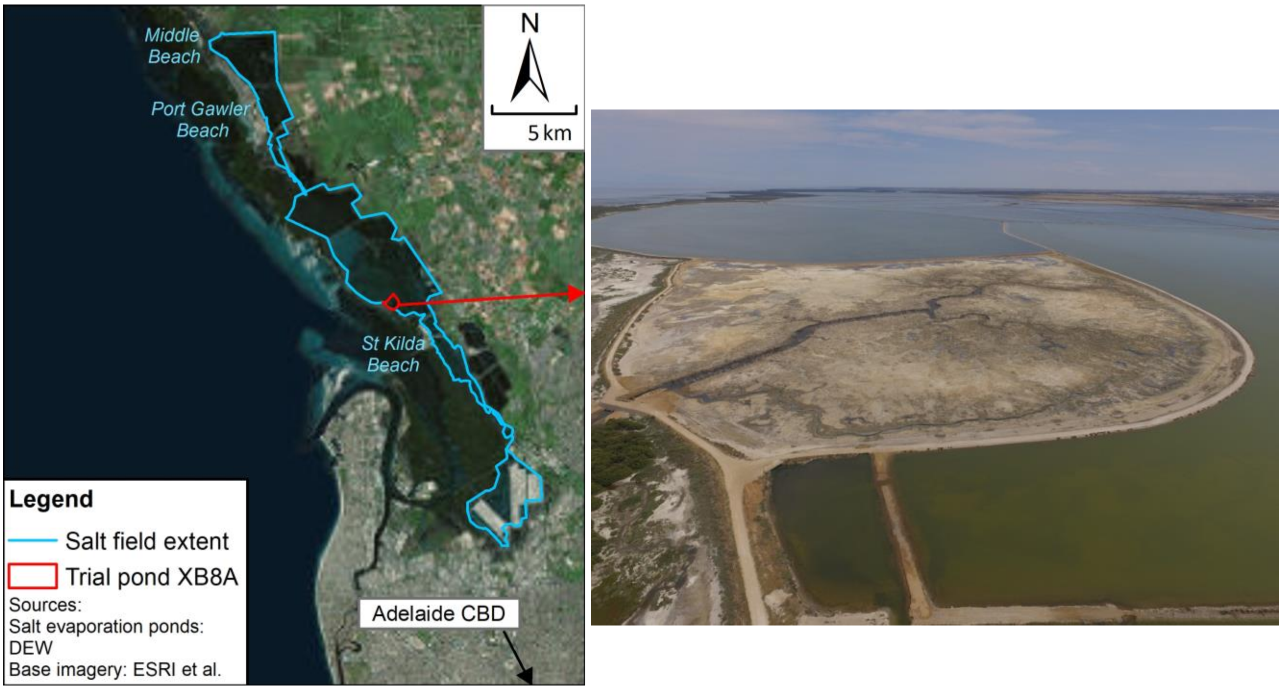

Figure 3. Location of the Dry Creek salt field north of Adelaide, South Australia, where solar evaporation ponds extend for ca. 30 km along the Gulf St Vincent coastline from Middle Beach in the north to Port Adelaide in the south. The trial pond, XB8A, is encircled in red and shown in an aerial photo after reconnection (photo DEW).

Pond XB8A was the original intake pond when operation of the salt field commenced in July 1937. Eighty years after operating as a salt field, the pond was reconnected to the Gulf of St Vincent on 28 July 2017 (Figure 4). In order to have control of water entering and exiting the pond (as required by regulatory authorities), tidal gate infrastructure was installed in the levee bank of Pond XB8A (Figure 5). The infrastructure consists of 4 x 1.2 m diameter x 10 m long polyethylene pipes and controllable tidal gates (AWMA i-gate). The system is powered by solar panels and can be controlled remotely. A multi-parameter water quality sensor (YSI EXO2) was installed in a conduit on the pond side of the gate, with level sensors also installed on both sides of the gate. Engineering design calculations (Tonkin Consulting 2016) were performed to provide the pipe sizing, orientation and elevation allowing for suitable water exchange within the typical tidal ranges. This design process was informed by surveys of the tidal creek and pond in the vicinity of the proposed structure, and tidal level logging.

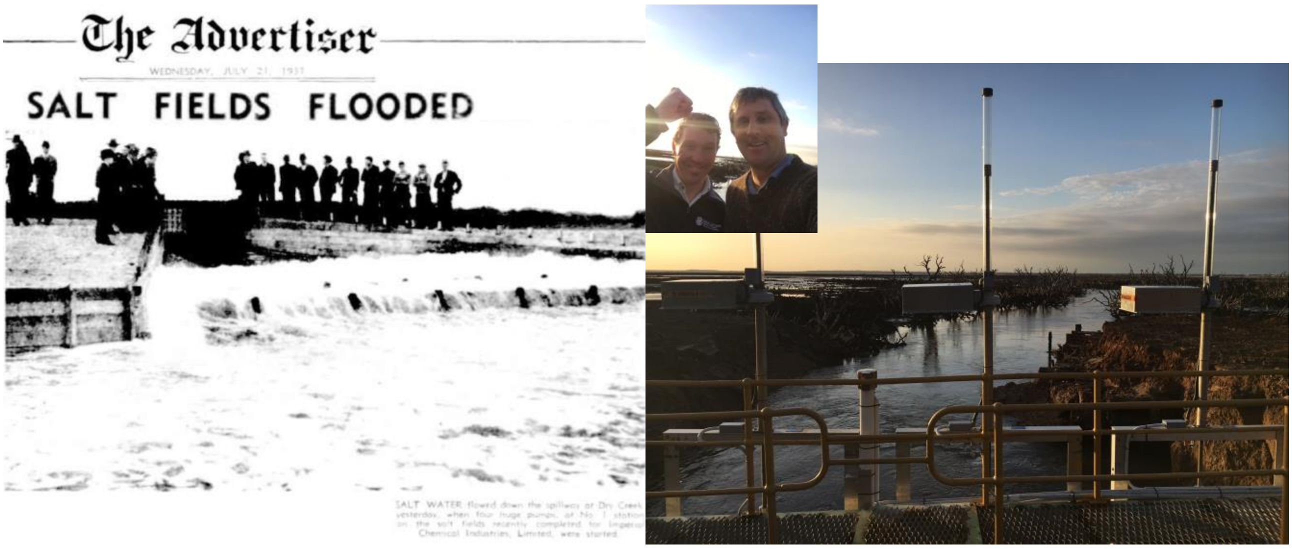

Figure 4. Historic opening ceremony of the Dry Creek salt field on 20 July 1937 at the No 1 pumping station, and celebration of tidal reconnection at the same location on 28 July 2017. Photos provided by Peter Bell and Luke Mosley.

Figure 5. Tidal gate infrastructure showing the controllable gates (left) and the polyethylene pipes connecting the trial pond to pumping creek and mangroves in Barker Inlet (right). Photos Luke Mosley and Sabine Dittmann.

2.2 Trial pond developments

The trial pond XB8A was reconnected at the end of July 2017. The re-introduction of tidal flow connected the central creek and associated side channels within the pond with the pumping creek flowing through a ca. 300 m wide band of mangrove and saltmarsh vegetation, located seaward of the pond (Figure 3b). The tidal reconnection successfully reduced the salinity inside the pond without causing any adverse effects on water quality inside or outside of the trial pond (Mosley et al. 2018).

The first saltmarsh plants established six months after reconnection inside the pond. Prior to reconnection, canopy branches of dead mangroves, which had lined the central creek when the salt pond was created, emerged above the water (Figure 6). The spatial extent of dead mangrove necromass became apparent once the pond was drained. Saltmarsh plants were first observed to emerge in November 2017 and established along the banks of the central creek over the following months (Figure 6).

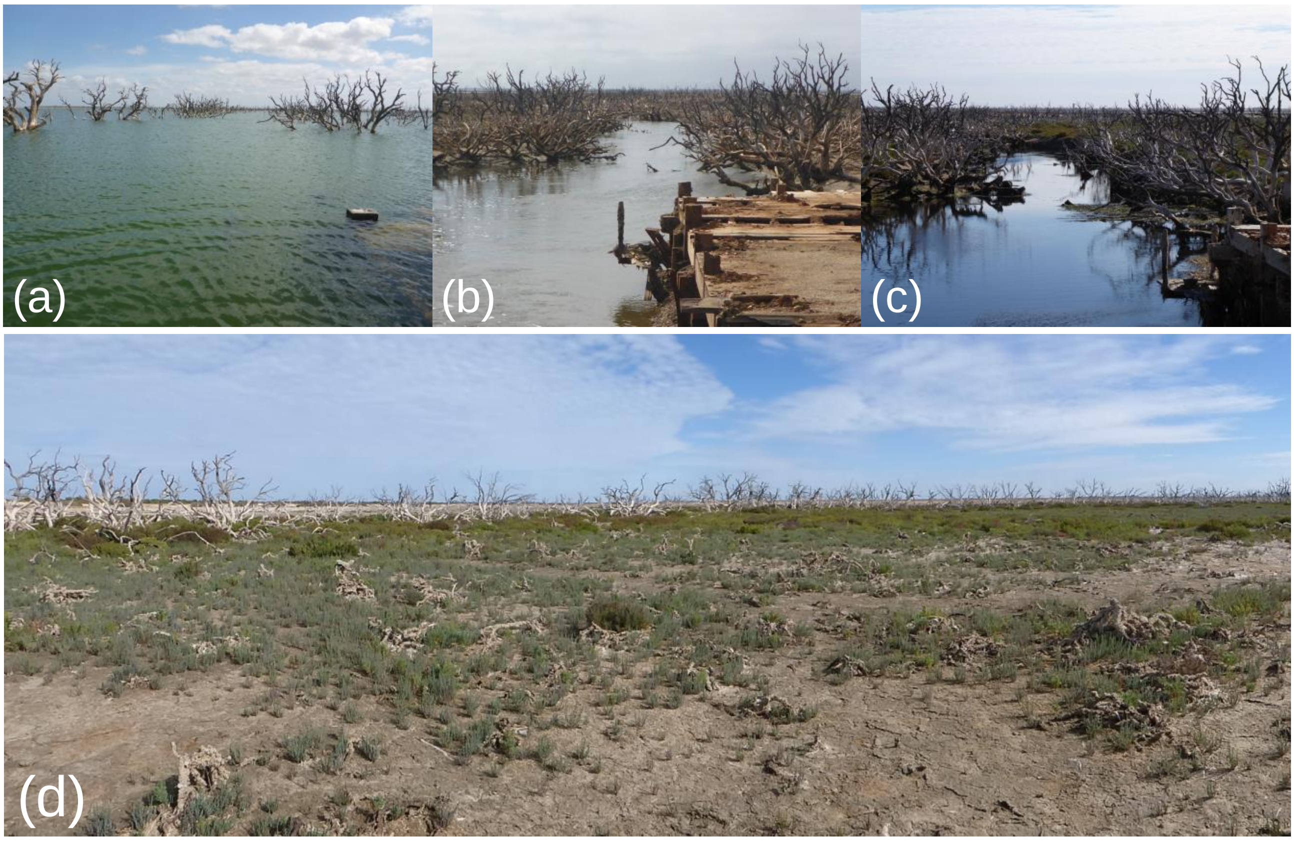

Figure 6. Photo sequence of the trial pond central creek (a) before reconnection in January 2017, (b) two months after tidal reconnection in September 2017, and (c) 20 months after reconnection in April 2019. (d) panorama view of revegetation inside the pond, April 2019. Photos Sabine Dittmann.

2.3 Study sites and stratification

The study was conducted at the Dry Creek Salt field along the samphire coast north of Adelaide, South Australia. The adjacent Gulf St Vincent is an inverse estuary, which experiences a mixed tidal pattern with meso-tidal range (ca. 2.8 m highest high tide) and slack water around neap tides (Bye and Kaempf 2008). The nearby Parafield airport receives an average annual rainfall of 451 mm per year (BoM 2019).

Mangrove forest, comprised solely of *Avicenna marina*, occurs seaward and extends along tidal channels, while areas of saltmarsh vegetation occur landward at relatively higher elevations (Harbison 2008; Fotheringham and Coleman 2008). Seagrasses grow within tidal channels in the mangroves and further into the adjacent gulf (Bryars et al. 2008).

Investigations were carried out at three sites, within the trial pond (TP), immediately outside the pond (adjacent reference area AR), and in a reference area (RA). The adjacent reference area was characterised by the pump creek, with dense mangrove, and saltmarsh communities which showed some affect from possible leakage of saline water from the salt field, as well as accumulation of litter and seagrass detritus. The reference area, located approximately one kilometre from the trial pond, was relatively intact. It was located at a similar distance from the low water line as the trial pond, but without any seaward barriers, thus representing a habitat and plant community which could be expected to occur in the trial pond if it had not been impounded. The vegetation at the lower elevations of the reference area was dominated by *Sarcocornia quinqueflora*, associated with *Tecticornia arbuscula* and small, scattered *Avicenna marina* mangrove growing on muddy soils. Shell grit cheniers form higher elevations and are covered by dense shrublands composed of *Maireana oppositifolia* and *Wilsonia humilis*. Samples were also taken from a nearby control pond (CP, pond

XB8) which was still flooded with hypersaline water to obtain data for the business-as-usual baseline conditions.

Data were gathered for three main elevation strata characterised by particular associated plant communities in an approach aligned with the VCS method VM0033 for project areas which are not homogenous (Emmer et al. 2015) (Table 2, Figure 7). Stratification by elevation levels also allowed modelling of carbon sequestration for several scenarios of reconnection of further ponds, over longer timeframes, and under sea level rise scenarios. Such assessment and modelling will be useful for considerations of a possible grouped blue carbon project and development of any future ERF methodology relevant to tidal reconnection.

We applied stratification of the project area into three main elevation levels which corresponded to particular vegetation classes, based on previous saltmarsh transect and elevation data from nearby Torrens Island (Fotheringham 1994, 2016). The strata used also followed classifications and predictive vegetation modelling by Herpich et al. (2017). Elevation was obtained from digital elevation models (DEMs) derived from high resolution aerial imagery for the trial pond, and from LiDAR and bathymetric surveys for the wider coastal area of the Samphire Coast and salt ponds. As the quality of the various DEM sources varied, elevations were ground-truthed by differential GPS at the three sites (Figure 8). The actual vegetation present at locations was also taken into consideration when defining strata for sampling sites, especially as the upper and lower elevation boundaries for many plants overlapped. In some project parts, the 'Mangrove-low marsh' stratum was divided into sub-strata, and mangrove forest only was sampled separately, in particular to obtain habitat specific carbon data (Figure 8).

Table 2: Strata delineations with prominent vegetation and elevation boundaries, as well as minimum and maximum elevations where some of the strata characteristic vegetation were recorded.

STRATA	VEGETATION	ELEVATION (mAHD)			
		LOWER BOUNDARY	UPPER BOUNDARY	MIN	MAX
Mangrove – low marsh (MLM)	*Avicennia marina, Sarcocornia quinqueflora, Suaeda australis, Tecticornia arbuscula*	<0.6	0.97	<0.6	1.19
Tidal saltmarsh (TSM)	*Sarcocornia quinqueflora, Tecticornia arbuscula, Suaeda australis Wilsonia humilis Maireana oppositifolia Avicennia marina*	0.97	1.35	0.89	1.59
Supra-tidal saltmarsh (SSM)	*Maireana oppositifolia, Tecticornia arbuscula Atriplex paludosa Frankenia pauciflora*	1.35	2.1	1.34	2.20

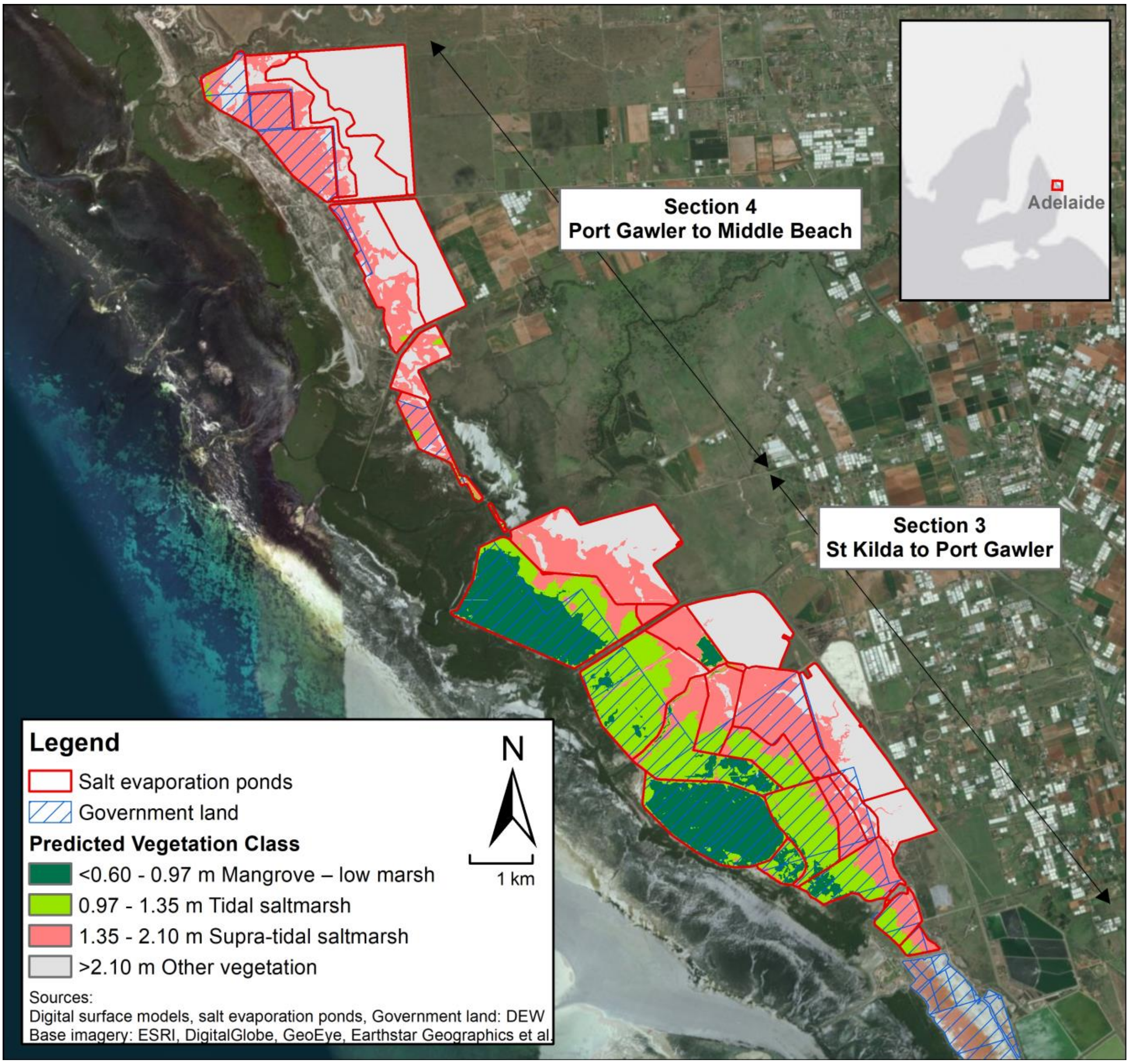

Figure 7. Map of two northern sections of the Dry Creek salt field showing the three strata ('Mangrove-low marsh' elevation 0.6–0.97 m, 'Tidal saltmarsh' elevation 0.97–1.35 m, 'Supra-tidal saltmarsh' 1.35–2.1 m) based on bathymetry estimates of submerged ponds. The map further indicates the land ownership, with Crown land outlined in blue.

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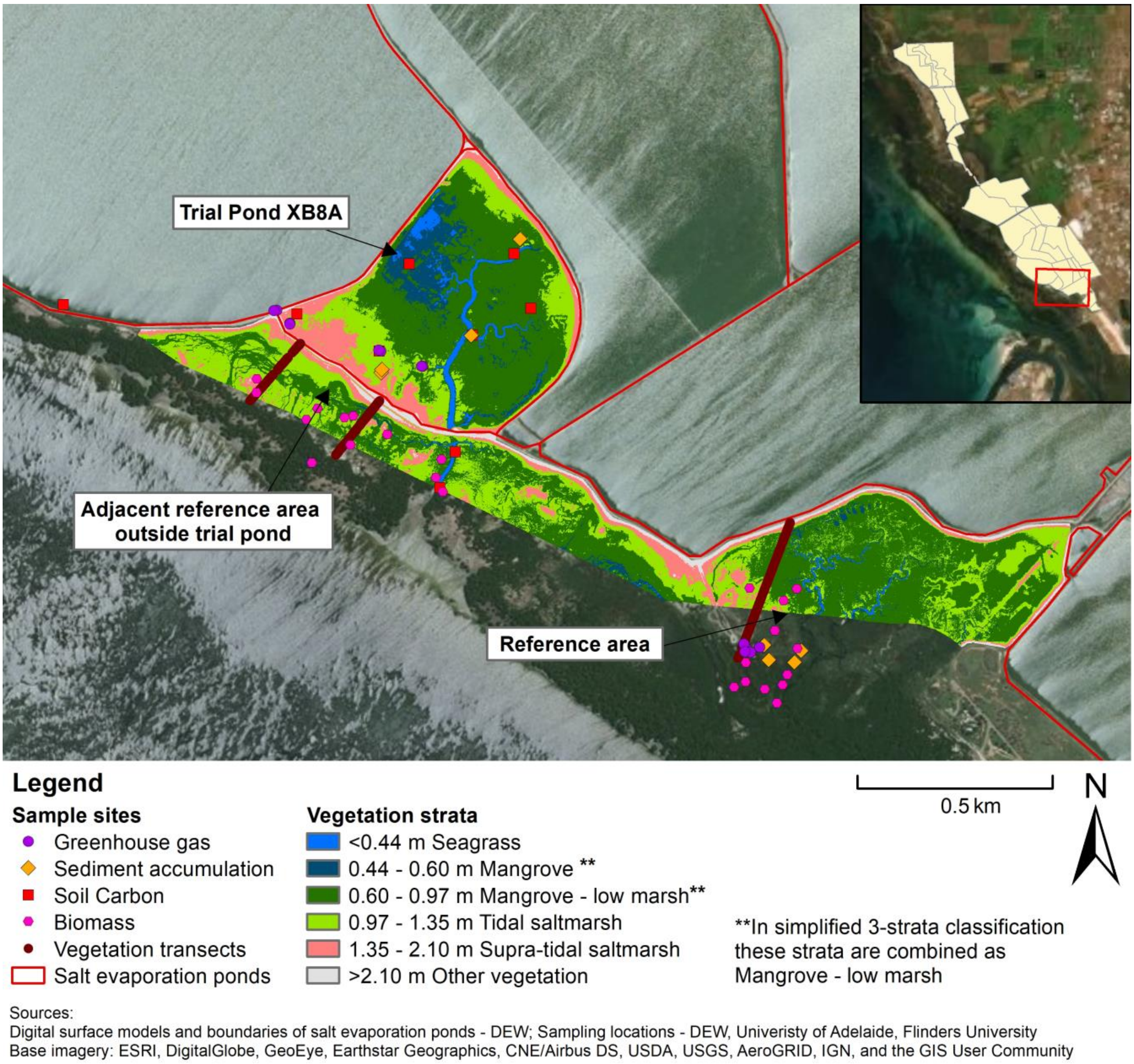

Figure 8. Map of sampling locations for various carbon pool measurements at the study sites within the trial pond, the control pond to the north, adjacent reference area outside the pond and reference area near St Kilda.

3 Carbon dynamics and sequestration

To ascertain whether tidal reconnection will lead to an increase in carbon sequestration, changes in the carbon pools were recorded over time inside the pond and reference sites. Sediments and vegetation were the main carbon pools considered, and the flux of GHG was also determined at the field sites. This chapter provides an overview on the carbon stock assessments. The data are used together with data from associated local studies in an accompanying proof of concept report for tidal reconnection (Dittmann et al. 2019). Methods used for determining carbon stocks in sediments and biomass followed the manual for blue carbon assessments (Howard et al. 2014), the VCS methodology VM0033 and the AR-Tool 14 as applicable and possible with local conditions and available data.

3.1 Carbon stocks and dynamics in sediments

3.1.1 METHODS

To follow changes in the sediment carbon stocks of the trial pond, samples were taken after draining of the XB8A pond in September 2017 (approx. two months after tidal reconnection), April 2018 (approx. nine months after tidal reconnection) and October 2018 (approx. 15 months after tidal reconnection). Samples were also taken from an adjacent pond (XB8) to assess changes in soil carbon stock over time under baseline condition (i.e. pond submerged by hypersaline water). Prior to reconnection, samples inside the trial pond were taken through a further project (Mosley et al. 2018) in May 2017. Soil carbon contents of the reference area and tidal creek were also taken in conjunction with the determination of sediment accumulation rates (see section 3.2).

Samples were collected via a Russian-D auger, pushed into the soil manually until refusal (typically around 50 cm). Total organic carbon and inorganic carbon were analysed via dry combustion and infrared detection before and after acid treatment (to remove/account for inorganic carbon component). Soil bulk density was determined by inserting a soil volume ring of known volume on vertical soil surface and extracting the soil contained in the ring, then air drying the soil and weighing the dried soil to produce bulk density (mass/volume).

Soil carbon stocks (SOC_{stock}) in mass per unit area (t CO_2e ha^{-1}) over a one metre depth plane were calculated via the VCS VM033 method (adapted[1] from equation 80):

$$SOC_{stock} = 44/12\ x\ \sum_{i=1}^{Ndepth}\left(C_{SOC\%,sample}\ x\ BD\ x\ Thickness\ \right) \quad (1)$$

where 44/12 is the ratio of molecular weight of CO_2 to carbon; N_{depth} is the number of soil horizons, based on subdivisions of soil cores; $C_{SOC\%,sample}$ is the total soil organic carbon of the sample (as determined in laboratory, %); BD is the bulk density, as determined in laboratory (g cm^{-3}), and thickness of the soil horizon (cm); and 100 is a unit conversion factor. As the soil cores extended to different depth (30–85 cm at most sites), the SOC_{stock} at each site was calculated to 30 cm depth for comparison across sites and times.

[1] Note a mistake was noted in the VCS Method equation 80, which is in the process of being corrected.

Carbon ($^{13}C/^{12}C$) and nitrogen ($^{15}N/^{14}N$) isotope ratios on selected sieved and freeze dried soil and seagrass samples were analysed by continuous-flow stable isotope ratio mass spectrometry using a Europa Scientific ANCA-SL elemental analyser coupled to a Geo 20-20 mass spectrometer at the CSIRO Waite Campus analytical facilities. Isotopic signatures were expressed in terms of delta (δ) values, which are in parts per thousand (‰) relative to a standard; for C this was Pee Dee limestone, and for N this was the nitrogen gas in the atmosphere.

3.1.2 RESULTS

The content of organic carbon in soils inside the trial pond area was highly variable (2–15%; Figure 9), likely reflecting the historical heterogeneity within the pond in terms of previous vegetation, elevations, and sediment conditions. The variability was also high between the different depth layers across the sampling sites. There were high concentrations of inorganic carbon (1–5%) in the soil, which is likely due to calcium carbonate precipitation from the hypersaline pond water and shell grit material.

There appears to be loss of organic C occurring at some sites, for example the well-drained XB8-04 site near the main tidal channel in the centre of the pond (XB8A-04). This may be due to aerobic mineralisation of organic matter. At some sites there was gain of inorganic C, for example site XB8A-01 at the margins of the pond which received irregular tidal flushing. Increases in inorganic C may be due to ongoing carbonate precipitation during evaporation of seepage water at this site.

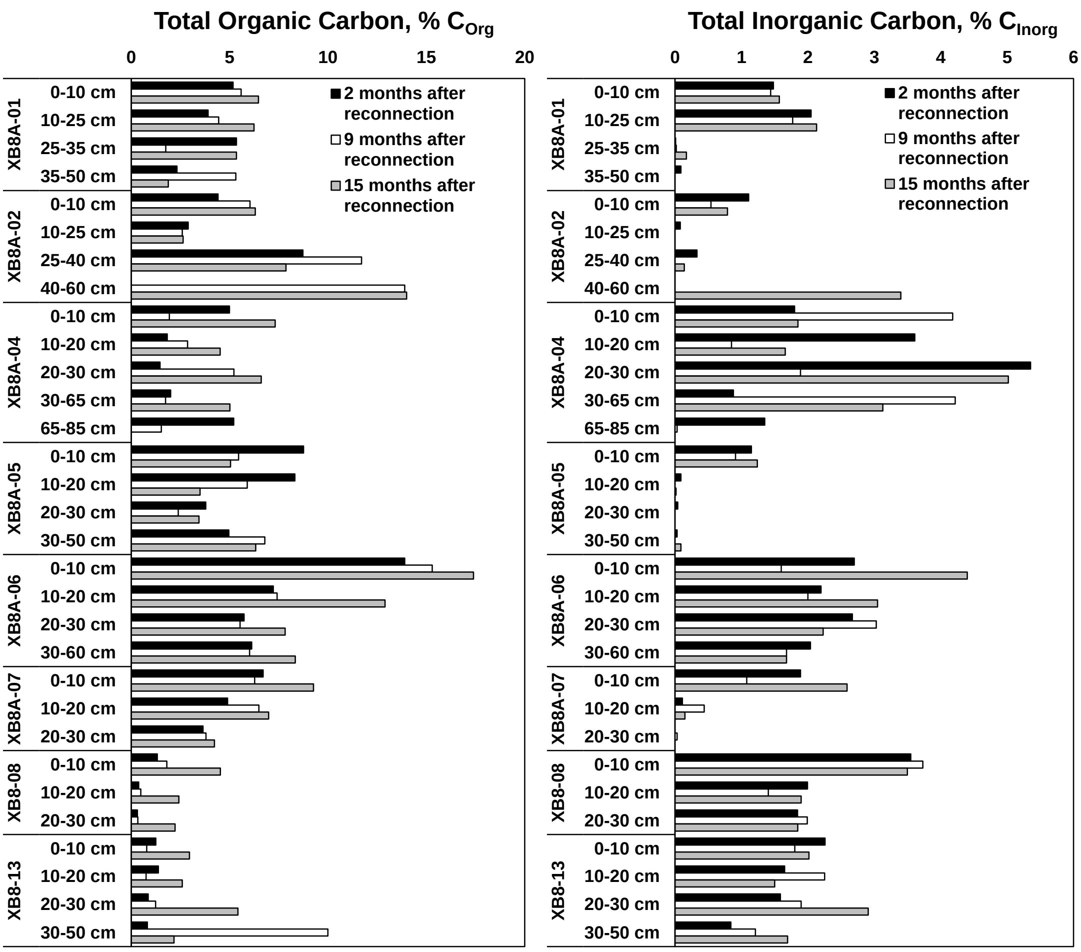

Figure 9. Total organic and inorganic carbon in the Dry Creek soil 2, 9 and 15 months after tidal reconnection.

The C stocks in the top 30 cm of the soil of the tidal trial pond for the three sampling occasions after tidal reconnection, are shown in Figure 10. An average net gain of 51 ± 28 t CO_2e ha^{-1} of soil organic carbon stock (for 30 cm soil layer) has occurred in the trial pond following 9 months of tidal reconnection, which became more variable after 15 months (46 ± 57 t CO_2e ha^{-1}). This variability was due to a decline in C stocks at the 'Supra-tidal saltmarsh' stratum inside the pond, whereas a continued increase in C stocks occurred at the 'Mangrove-low marsh' stratum, which covers the greatest area of the pond. The most rapid increase to a more steady C stock was recorded at the tidal marsh stratum site (Table 3). For the project area of the trial pond, the soil carbon values adjusted for the area of the respective strata reflect the increase and continued high levels of carbon stocks, thus no loss of soil carbon occurred due to tidal reconnection (Appendix A, Table A.1).

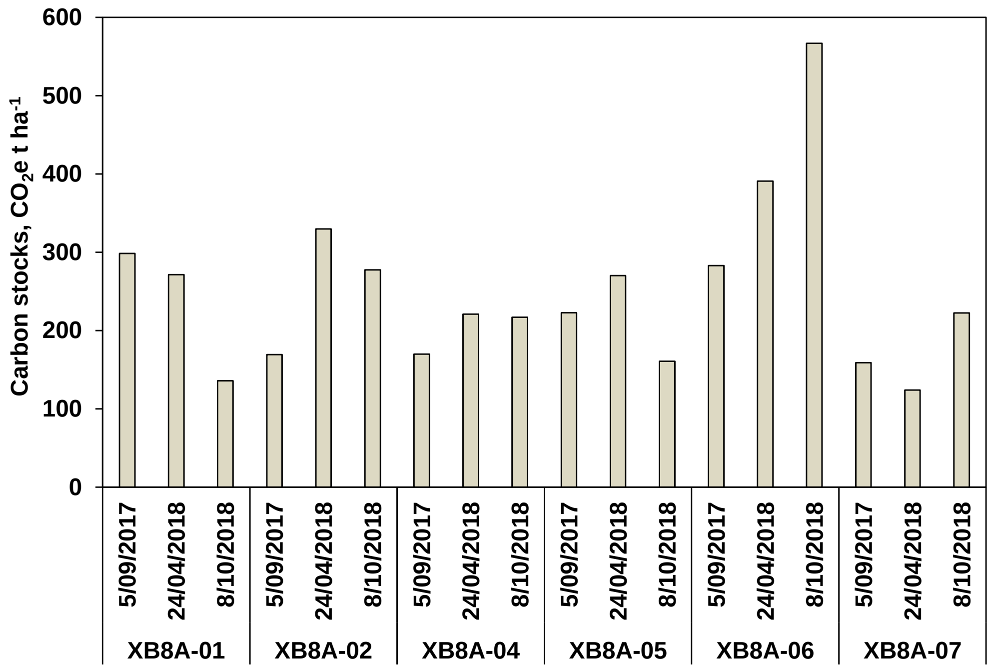

Figure 10 .Soil organic carbon stocks (CO_2 equivalents) at the Dry Creek tidal restoration trial site. Stocks are calculated to a 30 cm depth.

Table 3: Average (± SE) soil organic carbon stock (t CO_2e ha^{-1} in a 30 cm layer) for all sampling sites within the trial pond, and by strata, over three sampling dates after tidal reconnection. Of the six sites within the pond, four were in the 'Mangrove-low marsh' stratum, and one each in the other strata.

	t CO_2e ha^{-1}								
	5/09/2017			24/04/2018			8/10/2018		
TRIAL POND SITES/STRATA	MEAN	±	SE	MEAN	±	SE	MEAN	±	SE
All sampling sites	217	±	23	268	±	34	263	±	58
Mangrove – low marsh	209	±	20	251	±	39	292	±	66
Tidal saltmarsh	169			330			277		
Supra-tidal saltmarsh	298			271			136		

Given the minimal amount of revegetation at most sampling sites after 15 months of tidal restoration, the reasons for the net C stock gain (particularly in 0-10 cm layer, Figure 9) required further investigation. It was hypothesised to be due to a large amount of seagrass material (wrack) that had flowed in through the tidal gate, and been deposited and decomposed on the pond sediment surface (i.e. allochtonous C). Carbon and nitrogen isotope measurements supported this hypothesis with similar ratios in fresh seagrass compared to the 0–5 cm soil layer (Figure 11).

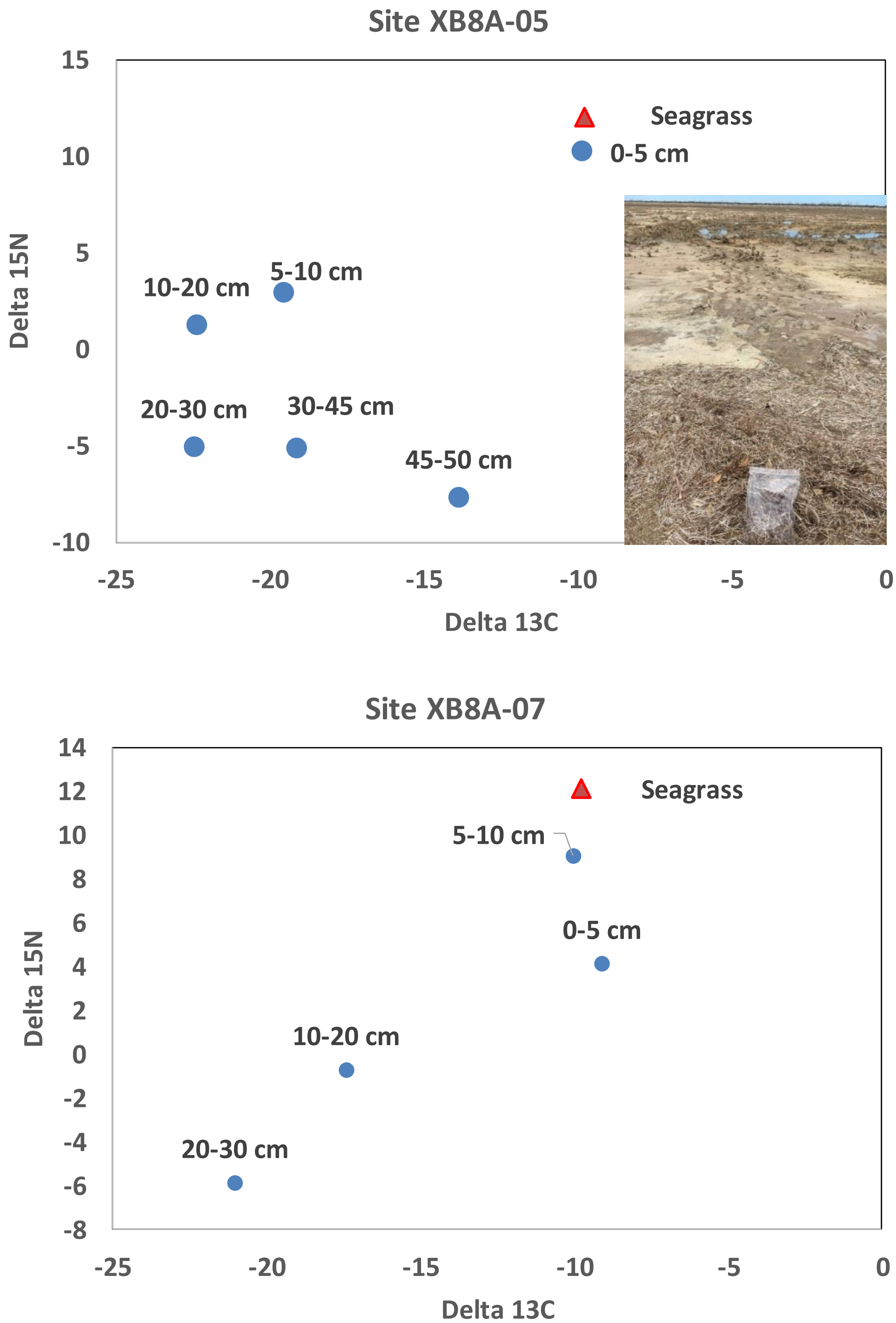

Figure 11. Carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) ratios from soil samples (5 cm layers) and seagrass material (wrack, see photo inset) collected at two sites in the trial pond.

3.2 Sediment elevation changes and accumulation rates

3.2.1 METHODS

Sediment accumulation rates were measured to determine sequestration and were obtained in two ways (see also section 3.3). Two rod surface-elevation tables (RSET) with a shallow benchmark (Cahoon et al. 2002) were installed, suitable for the conditions within the pond. At each time of sampling, nine rods were lowered to the sediment surface and the distance between the rod and the surface measured. The RSET was then rotated 180° and then process repeated on the other side (Figure 12).

Figure 12. Photo of a sediment elevation table in the trial pond.

3.2.2 RESULTS

The average accumulation rate at XB8A-02 was estimated to be 2.5 mm yr^{-1} and 0 mm yr^{-1} at XB8A-07, but there was large variability between pins on the rod (Table 4). There was a sediment inflow from the coastal water and sediment movement observed within the pond (e.g. deepening of channels), and salt mineral dissolution also occurred. The apparent sediment elevation increase between May and October 2018 at XB8A-02 ('Tidal saltmarsh' stratum) was not accompanied by further increase in soil C stocks, whereas soil C stock increased at XB8A-07 (Mangrove-low marsh' stratum) with no change in sediment elevation (see section 3.1). Further RSET data over a longer time period are required.

Table 4: Sediment elevation table results from the trial pond. The change (in mm) is relative to the installation date.

		change in mm		rate (mm yr^{-1})	
Site		May-2018	Oct-2018	May-2018	Oct-2018
XB8A-02	Average	1.2	3.1	1.6	2.5
	SD	*7.9*	*7.9*	*10.6*	*6.3*
XB8A-07	Average	-2.1	0.1	-2.8	0.0
	SD	*6.4*	*8.1*	*8.6*	*6.5*

3.3 Long term sedimentation and carbon sequestration rates

An understanding of the longer-term sediment and carbon accumulation rates is important to determine the likely capacity of the soil to sequester carbon as the system recovers.

3.3.1 METHODS

To determine longer term sediment accumulation rates, soil cores were collected for analysis of excess ^{210}Pb ($^{210}Pb_{ex}$) and ^{137}Cs from six locations in pond XB8A and the adjacent reference area. The sampling sites were distributed across the three vegetation strata, with 5 cm horizons sampled in 40 cm depth profiles. Additional soil samples were collected for determining organic carbon (5 cm intervals) and bulk density (10 cm intervals) using the methods described above.

Excess ^{210}Pb refers to the amount of ^{210}Pb that has accumulated from atmospheric deposition as opposed to *in situ* ^{210}Pb production, which is referred to as supported ^{210}Pb. ^{210}Pb is produced from the radioactive decay of ^{222}Rn and has a half-life of 22.26 years. Radon is a gas and ^{210}Pb is of course a solid which when formed from the radon gas falls out of the atmosphere and accumulates on the ground surface. *In situ* production (in the sediment) of ^{210}Pb is calculated from the concentration of ^{226}Ra which decays to produce ^{222}Rn. To check whether the excess ^{210}Pb content in the sediment column is due to recent sediment accumulation, ^{137}Cs data were also obtained. If the sedimentary system has had a simple accumulation history, a spike in concentration of ^{137}Cs in the sediment column is assumed to be from fallout from above ground nuclear testing, which peaked in the middle 1960's (1963-64 in the northern hemisphere).

Having determined the sediment column has a simple accumulation history, a sediment accumulation model can be applied, such as the Constant Flux Constant Sedimentation (CFCS) or the Constant Rate of Supply (CRS) model (see Appendix A.2 for detail on model calculation). The isotope dating analyses, and determination of C_i for the models, were performed at the Australian National Science and Technology Organisation (ANSTO). There was a problem with generally low ^{210}Pb activities in the sediment/soil profiles, which did not allow for all the profiles sampled and analysed to be utilised. Bulk samples were taken and were not sieved and high concentrations in some profiles of shell debris probably resulted in low $^{210}Pb_{ex}$ activities. Two of the profiles (DC-S3 and DC-S7) were analysed a second time by the more sensitive alpha spectrometry at ANSTO labs.

Carbon sequestration rates (C_{seq}; g C m^{-2} yr^{-1}) were calculated in each sampled soil layer using:

$$C_{seq} = Sed \times BD \times C_{SOCconc} \qquad (2)$$

where *Sed* is the sedimentation rate supplied by the CRS or CFCS ^{210}Pb methods (cm yr^{-1}), BD is the bulk density (g-soil cm^{-3}) throughout the dated (past 50 years) portion of the soil core profile, and $C_{SOCconc}$ (g-C g-soil^{-1}, equivalent to SOC% x 10) is the soil carbon concentration measured in the soil layer.

Utilising the sedimentation rates determined from the Constant Rate of Supply (CRS) age model, the carbon sequestration rates of the four representation profiles were calculated (Appendix A, Figure A.2). These calculations applied dry bulk density, five centimetre depth intervals, and total carbon and organic carbon analyses which yield carbon sequestration in the form of g m^{-2} yr^{-1}.

3.3.2 RESULTS

Sedimentation and C-sequestration rates for the sites where reliable dates could be obtained are shown in Table 5. In general, the sedimentation and C sequestration rates are comparable with those obtained in a previous study by Dittmann et al. (2016) in the general vicinity (Middle Beach, Swan Alley, Little Para).

The DC-S1 site, situated on the historical 'Mangrove-low marsh' in the trial pond, had the highest sedimentation rates of 13 mm yr^{-1} (CRS) and 9.8 mm yr^{-1} (CFCS). ^{210}Pb activities were relatively homogenous down to 40 cm depth, which is probably due to sediment mixing by bioturbation. Channel bank or levee ridge environments would be expected to have relatively high sedimentation rates due to suspended sediment loads from tidal inflows. However, over the last few decades the site has been a salt pond so the tidal channel was not operating as a source of allochtonous sediment. The pond had stable water levels and low through-flow velocities, both of which are conducive sedimentation conditions. Sources of sediment within the pond could include organic matter produced by algae in water and benthic mats, mineral precipitation reactions, and dust.

The DC-S2B site 'Tidal saltmarsh' is more representative of a large part of the pond. The topmost five centimetre interval consisted of loose recently disturbed gypsum and shell hash and had $^{210}Pb_{ex}$ activities below detection limit, and a peat layer at 30 cm with very low $^{210}Pb_{ex}$ activities, so both were omitted from the age model calculations. Again, the top interval was omitted from the age model calculations. Caesium 137 activities were below detection limit except for one interval at 20–25 cm depth. Sedimentation rates were different when calculated using the CRS (2.4 mm yr^{-1}) and CFCS (7.5 mm yr^{-1}) models.

The DC-S7 'Mangrove-low marsh' site is representative of a large part of the reference area. All five upper profile intervals had moderate to low $^{210}Pb_{ex}$ activities as determined by alpha spectrometry and no intervals were omitted in the age model calculations. Sedimentation rates of 2.2 mm yr^{-1} (CRS and CFCS models) were low which is consistent with the RSET measurements (Table 5).

The DC-S3 'Mangrove-low marsh' site is also representative of a large part of the reference area. The upper five profile intervals had moderate to high $^{210}Pb_{ex}$ activity as determined by alpha spectrometry and no intervals were omitted in the age model calculations. Sedimentation rates of 1.5 mm yr^{-1} (CRS) and 2.4 mm yr^{-1} (CFCS) were low and similar to Site DC-S7 and the RSET measurements.

The two pond sites (trial and control pond) had very different rates of carbon sequestration (Table 5). Site DC-S1 'Mangrove-low marsh', with its high sedimentation rate and moderate total carbon and organic carbon concentrations had moderate to high carbon sequestration values. Site DC-S2B 'Tidal saltmarsh', with its low sedimentation rate and high carbon concentrations, yields moderate to low carbon sequestration values. The two reference sites (DC-S7 and DC-S3, 'Mangrove-low marsh') had low sedimentation rate and moderate total carbon concentrations, yielding much lower values of carbon sequestration.

Table 5: Sedimentation rate and average carbon sequestration rates (as g C and in CO_2 equivalents) for organic carbon for the trial pond (TP; DC-SC1 and DC-SC2B) and reference area (RA; DC-S3 and DC-S7) calculated using the constant rate of supply (CRS) and constant flux constant sedimentation (CFCS) sedimentation models.

			Sedimentation rate		C sequestration rate			
			mm yr^{-1}		g C m^{-2} yr^{-1}		t CO_2e ha^{-1} yr^{-1}	
Sample	Site	Strata	CRS model	CFCS model	CRS model	CFCS model	CRS model	CFCS model
DC-S1	TP	Mangrove-low marsh	13	9.8	671	506	24.6	18.5
DC-S2B	TP	Tidal saltmarsh	2.4	7.5	195	609	7.1	22.3
DC-S3	RA	Mangrove-low marsh	1.5	2.4	81.5	130.4	3.0	4.8
DC-S7	RA	Mangrove-low marsh	2.2	2.2	42.4	42.4	1.6	1.6

3.4 Greenhouse gas emissions

Changes in GHG emissions following reconnection of the pond are an additional component of carbon abatement in the course of the restoration. Thus, GHG emissions were measured to provide respective data for carbon accounting for the site.

3.4.1 METHODS

Greenhouse gas emissions were monitored on several occasions at three different sites: in the trial pond, the adjacent control pond representing the salt pond baseline, and at the reference area of an established natural tidal marsh. For the trial pond and the reference sites, different elevation strata were chosen, primarily including 'Mangrove-low marsh', 'Tidal saltmarsh', and 'Supra-tidal saltmarsh' (Figure 8).

Samples were collected using fixed and floating chambers (Appendix A, Figure A.3) connected to a gas sampling bag at the first two surveys (September and November 2017). At the survey in May 2018, a LICOR LI-8000 CO_2 gas analyser was used for CO_2 measurements (Appendix A, Table A.2). Sub-samples were collected in gas-tight syringes from the bags and injected through a rubber septum into evacuated gas-tight glass vials. At the time of analysis, an aliquot of gas was removed from the vial and the concentration of CO_2, CH_4 and N_2O measured by gas chromatography. The flux across the sediment or water surface was calculated from the change in gas concentration, time between measurements, and the surface area of the chamber.

A carbon dioxide equivalent (CO_2e) was calculated for nitrous oxide and methane emissions. This is a metric measure used to compare the emissions from various GHG on the basis of their global-warming potential by converting amounts of other gasses to the equivalent amount of carbon dioxide with the same global warming potential (GWP). The GWP for methane is 36 and for nitrous oxide 298 and their CO_2e was derived by multiplying the GWP by the measured fluxes of these gases.

3.4.2 RESULTS AND DISCUSSION

Of the three GHG measured on the first two sampling occasions, CO_2 accounted for 96% of the total GHG fluxes on average (Figure 13). Emissions of N_2O and CH_4 were thus negligible compared to CO_2 fluxes. The minimal methane flux is consistent with international findings of minimal flux above a salinity of 18 due to sufficient sulfate being present (Poffenbarger et al. 2011), which is a more thermodynamically favourable electron acceptor, and hence methane flux is considered to be *de minimis* for the purposes of carbon accounting. Emissions of methane were also negligible inside a nearby mangrove forest (Hill, 2018).

The three field measurement campaigns were taken in early spring, late spring and late autumn. An average of them likely provides a fair approximation for the annual GHG fluxes, if the effect of possible diurnal variation is neglected. It is reported that CH_4 fluxes in saltmarsh environments do not experience diurnal variation (Bartlett et al. 1987), but it is unknown if this also applies to N_2O and CO_2. With these assumptions, the annual GHG fluxes were estimated to be 3.5 ± 2.2 t CO_2e $ha^{-1} yr^{-1}$ at the tidal reconnected pond site, in comparison to 2.2 ± 1.1 t CO_2e $ha^{-1} yr^{-1}$ at the reference site, and 0.6 ± 0.5 t CO_2e $ha^{-1} yr^{-1}$ at the control pond (Table 6). The seemingly higher average flux for the trial pond was largely due to a single high reading from the supra-tidal sampling location, and no significant differences were detected for either of the three gasses nor the sum of the GHG fluxes between the three sites or sampling dates (two-way PERMANOVA, P>0.05). The global averaged coastal wetland net ecosystem production is 2.08 t CO_2e $ha^{-1} yr^{-1}$ (Lu et al. 2017), and with further revegetation, the trial pond may become a carbon sink.

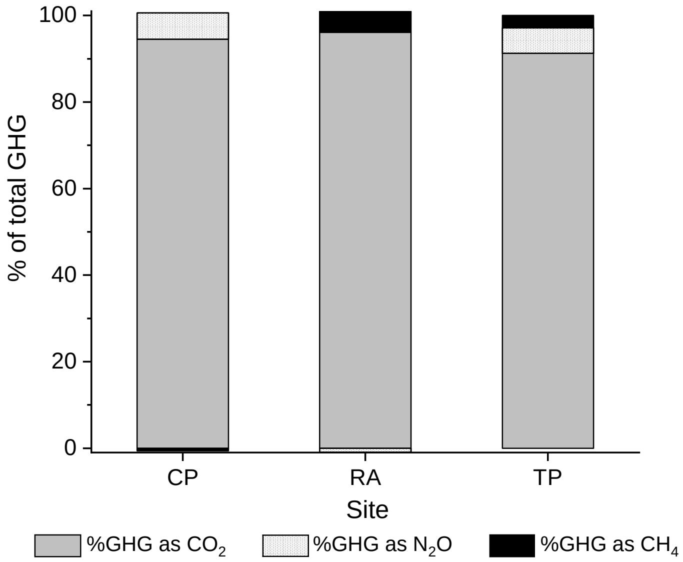

Figure 13. Percentage contribution of the main GHG (CO_2, N_2O and CH_4) to the total GHG flux across the sampling sites, based on three sampling occasions. CP = Control pond, RA = Reference area, TP = Trial pond.

The GHG fluxes were highly variable across the sampling sites and times (Figure 14), with no clear pattern emerging about differences in the sink or source of emissions from sediments (trial and reference sites) or surface water (control pond site) (Appendix, Figure A.4). According to Bartlett et al. (1987) and Allen et al. (2011), the GHG fluxes in coastal wetlands peak in summer and autumn, and our measurements could indicate seasonal variation. The maximum air temperatures measured at the Adelaide Airport were 23.2°C, 22.8°C, and 17.7°C, respectively. Given the possible seasonal variation of GHG fluxes, the estimates for annual GHG fluxes have to be interpreted with caution.

Table 6: Fluxes (average ± SE) of greenhouse gas emissions expressed in t CO_2e ha^{-1} yr^{-1} at each of the study sites, based on three surveys in 2017–18.

	EMISSIONS t CO_2e ha^{-1} yr^{-1}											
	CO_2			N_2O			CH_4			SUM GHG		
SITES	MEAN	±	SE	MEAN	±	SE	MEAN	±	SE	MEAN	±	SE
Control pond	0.58	±	0.44	0.05	±	0.04	-0.004	±	0.01	0.61	±	0.45
Reference area	2.13	±	1.07	0.01	±	0.02	0.06	±	0.04	2.18	±	1.08
Trial pond	3.25	±	2.15	0.17	±	0.10	0.17	±	0.11	3.47	±	2.16

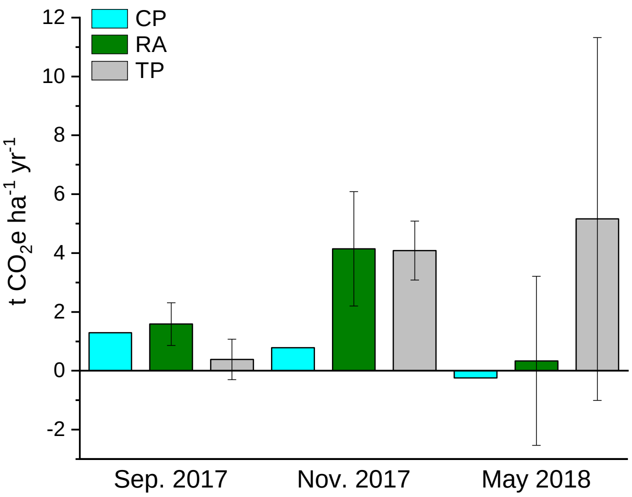

Figure 14. Fluxes of CO_2e (average ± SE) across the sampling sites at three sampling occasions. Note that the method of measurement changed between the first two occasions (September and November 2017) and the last measurement in May 2017. CP = Control pond, RA = Reference area, TP = Trial pond.

GHG fluxes were also highly variable across the elevation strata (Figure 15). The 'Tidal saltmarsh' stratum seems to have lower GHG fluxes, while the 'Supra-tidal saltmarsh' strata had higher, but also more variable GHG fluxes. Thus, strata specific patterns of GHG emissions did not emerge.

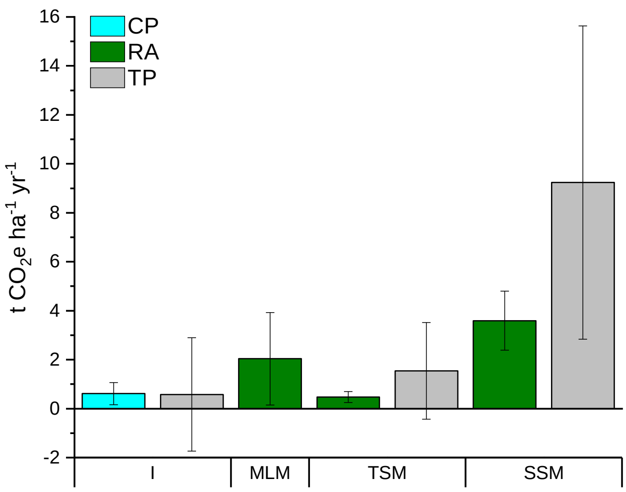

Figure 15. Fluxes of CO_2e (average ± SE) across the strata in the trial pond, control pond and reference area. The elevation strata were I = inundated, MLM = 'Mangrove-low saltmarsh', TSM = 'Tidal saltmarsh', and SSM = 'Supra-tidal saltmarsh'. CP = Control pond, RA = Reference area, TP = Trial pond.

3.5 Carbon stocks and sequestration in biomass

Carbon captured in the biomass, above (AGB) and below (BGB) ground, of mangrove and saltmarsh plants is a further relevant component of blue carbon stocks. It was expected that following tidal reconnection, the trial pond would be re-colonised by mangrove and saltmarsh, but that the vegetation will not reach a steady-state within the project timeframe (see chapter 4 for revegetation). The carbon pool in saltmarsh and mangrove vegetation was therefore quantified at two sites, adjacent to the trial pond and the reference area. Measurements were taken for three intertidal plant communities, *Avicennia marina* mangrove forest, *Sarcocornia quinqueflora* tidal saltmarsh and *Maireana oppositifolia* shrubland/saltmarsh, based on vegetation communities and elevation maps available at the start of the project to define strata (see section 2.3). For the 'Mangrove-low marsh' stratum, carbon data are presented separately for mangroves and saltmarsh vegetation, to allow more accurate estimates and forecasting of carbon sequestration potential with revegetation changes after tidal reconnection. The mean mangrove tree density in the reference areas was 3000 trees per hectare (± 518 SE), with a mean height of 3.8 m (± 0.1 m SE).

3.5.1 METHODS

The above and below-ground biomass (t ha^{-1}) and vegetation carbon pools (t C ha^{-1}) were assessed within four, 5×5 m plots per vegetation type and per site. Carbon pools included live plant material (above-ground), live roots (below-ground), pneumatophores and litter. Methods used followed Howard et al. (2014).

For mangrove, the above-ground biomass was estimated using an allometric relationship developed for *A. marina* by Clough et al. (1997). The relationship is based on stem (or butt) measurements taken close to ground level. Thus, the allometric relationship is adaptable for multi-stemmed trees or trees that have branching points close to ground level, which can make diameter-at-breast-height (dbh) measurements impracticable. Stem or butt measurements (as well as tree height) were recorded for all trees rooted within the 5×5 m plots.

The biomass of mangrove pneumatophores was estimated by counting the number of pneumatophores within five 0.25×0.25 m quadrats (sub-samples) per plot and multiplying the average density of pneumatophores at each plot by the average biomass of pneumatophores obtained for each site. The average biomass of pneumatophores was obtained by harvesting all pneumatophores that occurred within a subset of the 0.25×0.25 m quadrats. Samples of litter, where present, were also collected from the 0.25×0.25 m quadrats. Twigs with diameters of ≤2.5 cm were included in litter samples. Samples were rinsed over a 1 mm sieve to remove soil prior to being dried at 60°C until constant weight was reached.

For saltmarsh, above-ground biomass was determined for two saltmarsh communities. From each plot within the *S. quinqueflora* saltmarsh, two 0.25×0.25 m quadrats (subsamples) were harvested. For *M. oppositifolia* shrubland one, 1×1 m quadrat was harvested per plot. Samples consisted of *M. oppositifolia* in addition to *Wilsonia humilis* in some instances, and the biomass of the two species were quantified separately. *S. quinqueflora* samples were rinsed over a 1 mm sieve and samples from both communities were dried at 60°C until constant weight was reached.

Roots were sampled by taking soil cores to a maximum depth of 45 cm. Five soil cores, divided into 5 cm depth categories, were taken from each *S. quinqueflora* saltmarsh and *A. marina* plot using a peat corer (4.3 cm diameter). The peat corer was ineffective for sampling the shell grit sediment of the *M. oppositifolia* shrubland, thus a hand auger (6.9 cm diameter) was used, although no separate depth intervals could be taken. Samples were frozen at -4°C prior to processing. Samples were rinsed through a 4 mm sieve in order to remove large debris and collect large root material if present. Samples were further rinsed through a 1 mm sieve to retain finer root material, which were also floated in a water-filled container and decanted off to separate roots from mineral particles (e.g. shell grit). Efforts were made to remove dead roots and other organic material. The sieving process described was also conducted for material that passed through the 1 mm sieve and collected in a 0.5 mm sieve, however, consistently clean root samples were difficult to obtain for this super-fine fraction and it was not included in estimates of root biomass. The presence of super-fine root material indicates that root biomass is likely to be underestimated. Samples were dried at 60°C to a constant weight.

Carbon content of the AGB and BGB carbon pools was measured with a CHN analyser (Elementar Cube). Replicate values for vegetation, roots, litter and pneumatophores were obtained by analysing the sampled biomass at the plot level. In order to achieve homogenous samples for C analysis, vegetation, litter and downed wood biomass were mulched (Talon 3hp Garden Shredder) and subsamples ground to a fine powder (IKA M20 Universal Mill followed by IKA Tube Mill). Root samples from a given plot were bulked over depth categories and cores and all material was ground to a powder using the IKA mills. While efforts were made to remove all mineral material from root samples, low level contamination by shell-grit (calcium carbonate) was possible which could contribute to the carbon content of the samples. Therefore, a subsample of each root sample was ashed at 450°C for five hours and the ash was analysed for any remaining inorganic carbon. These values were used to derive the organic carbon content of the root samples. Finally, a total of sixteen pieces of downed wood ranging in size (2.7–10.0 cm diameter) and five randomly selected pneumatophores from each mangrove plot were ground for C content analysis.

Carbon pools for above-ground (AG) vegetation, below-ground (BG) roots, pneumatophores, and litter were estimated by multiplying biomass values by the respective carbon content fraction. Carbon content of *A. marina* AG vegetation was not measured and so a carbon conversion value of 0.47 used. Carbon content values for mangroves between 0.46–0.50 are suggested in the absence of local values (Howard et al. 2014) and a value of 0.47 was obtained for *A. marina* in the Sydney (NSW, Australia) region (Owers et al. 2018).

Differences in AGB, BGB and total vegetation biomass (AG+BG biomass) between the three vegetation types were tested using one-way ANOVA and Tukey post-hoc tests. Differences in carbon stocks (AG, BG and AG+BG) were analysed in the same way. All data were log-transformed to meet ANOVA assumptions of normality and equality of variance between groups. While transformations improved equality of variance, the assumption was not met for BG biomass (Levene's statistic = 3.56, p = 0.047) or for the BG carbon pool (Levene statistic = 4.49, p = 0.024).

3.5.2 RESULTS

Total plant biomass (AG+BG biomass) varied between vegetation types ($F_{2,21}$ = 93.23, p <0.001), with significant differences between all pairwise comparisons (Tukey p $\leq$0.008). Total biomass estimates were highest for *A. marina* (332.9 $\pm$ 39.01 t ha^{-1}), followed by *S. quinqueflora* (53.2 $\pm$ 10.06 t ha^{-1}) and *M. oppositifolia* (25.2 $\pm$ 1.58 t ha^{-1}) (Table 7). Above ground vegetation accounted for most of the total biomass for *A. marina* and *M. oppositifolia*, whereas BG root material accounted for most of the biomass of *S. quinqueflora*.

The AGB was significantly different between the three vegetation types ($F_{2,21}$ = 216.92; p <0.001) and all pairwise comparisons (Tukey p <0.018). As expected, *A. marina* had substantially greater AG biomass than the two saltmarsh communities, although for the latter, *M. oppositifolia* had more AG biomass than *S. quinqueflora* (Table 7). Six of the eight AG samples of *M. oppositifolia* shrubland included *Wilsonia humilis*, which accounted for an average of 10.2 $\pm$ 3.49% of the AG biomass (N = 8). Below-ground biomass also depended on vegetation type ($F_{2,21}$ = 87.81; p <0.001); however, the post-hoc tests showed that *A. marina* and *S. quinqueflora* had similar BGB (Tukey p = 0.605). These two vegetation types had significantly more BGB than *M. oppositifolia* (Tukey p <0.001) (Table 7).

Table 7: Vegetation biomass and carbon pools for three vegetation types. Values are presented for above-ground (AG) vegetation and below-ground (BG) roots, and for the data pooled over the AG (including pneumatophores for mangroves) and BG pools.

VEGETATION	STRATA	VEGETATION TYPE	AG VEGETATION			BG ROOTS			AG + BG		
			MEAN	±	SE	MEAN	±	SE	MEAN	±	SE
Biomass (t DW ha^{-1})	MLM	*Avicennia marina*	289.99	±	37.78	42.9	±	5.34	336.40	±	39.02
	MLM	*Sarcocornia quinqueflora*	14.03	±	1.13	39.16	±	9.10	53.18	±	10.06
	TSM	*Maireana oppositifolia*	22.25	±	1.48	2.99	±	0.22	25.24	±	1.58
C pools (t C ha^{-1})	MLM	*Avicennia marina*	136.3	±	17.75	18.61	±	2.27	156.50	±	18.22
	MLM	*Sarcocornia quinqueflora*	5.35	±	0.45	15.02	±	3.31	20.37	±	3.70
	TSM	*Maireana oppositifolia*	9.96	±	0.63	1.33	±	0.09	11.28	±	0.68
CO_2e (t C ha^{-1})	MLM	*Avicennia marina*	499.75	±	65.10	68.23	±	8.31	573.83	±	66.82
	MLM	*Sarcocornia quinqueflora*	19.63	±	1.65	55.06	±	12.15	74.69	±	13.57
	TSM	*Maireana oppositifolia*	36.50	±	2.31	4.87	±	0.34	41.38	±	2.50

The biomass of litter was higher at the adjacent reference area than the reference area (Table 8). Large amounts of litter typically occurred at the adjacent reference area, in the mangrove forest (*A. marina*) and the *M. oppositifolia* shrubland, due to accumulated seagrass detritus. Considerably less litter occurred in the low saltmarsh dominated by S. *quinqueflora* (Table 8). Litter was mostly composed of seagrass detritus.

Table 8: Litter biomass and carbon pools for three vegetation types at the adjacent area and reference area.

			ADJACENT AREA			REFERENCE AREA		
LITTER	STRATA	VEGETATION TYPE	MEAN	±	SE	MEAN	±	SE
Biomass (t DW ha^{-1})	MLM	*Avicennia marina*	11.18	±	4.21	1.70	±	0.30
	MLM	*Sarcocornia quinqueflora*	0.72	±	0.64	0.44	±	0.42
	TSM	*Maireana oppositifolia*	12.51	±	1.94	3.96	±	1.33
C pools (t C ha^{-1})	MLM	*Avicennia marina*	4.20	±	1.61	0.76	±	0.14
	MLM	*Sarcocornia quinqueflora*	0.26	±	0.24	0.16	±	0.15
	TSM	*Maireana oppositifolia*	5.29	±	0.7	1.75	±	0.59
CO_2e (t C ha^{-1})	MLM	*Avicennia marina*	15.41	±	5.89	2.79	±	0.51
	MLM	*Sarcocornia quinqueflora*	0.96	±	0.87	0.59	±	0.56
	TSM	*Maireana oppositifolia*	19.39	±	2.56	6.40	±	2.17

Carbon content values (% carbon) for the measured carbon pools ranged from 37% to 48% (overall average 42.85%, without litter 43.78%) (Table 9). The carbon content of root samples was adjusted for low levels of contamination by inorganic carbon, which averaged 0.54%.

Carbon pools of AG vegetation and BG roots combined (AG+BG C stock, Table 7) were estimated to be 156.5 ± 18.22, 20.4 ± 3.70 and 11.3 ± 0.68 t C ha^{-1} for *A. marina*, *S. quinqueflora* and *M. oppositifolia* vegetation respectively. The carbon pools were significantly different among the three vegetation types ($F_{2,21}$ = 102.20, p <0.001) for all pairwise comparisons (Tukey p ≤0.044).

Table 9: Percent carbon content of above- and belowground vegetation components, as well as dead plant material, based on assessments at the reference areas.

CARBON POOLS	VEGETATION TYPE	% CARBON	±	SE
Above-ground vegetation				
Pneumatophores	*Avicennia marina*	45.4	±	0.17
Sarcocornia quinqueflora	*Sarcocornia quinqueflora*	38.1	±	0.52
Maireana oppositifolia	*Maireana oppositifolia*	44.6	±	0.52
Wilsonia humilus	*Maireana oppositifolia*	47.2	±	0.33
Below-ground vegetation				
Roots	*Avicennia marina*	43.6	±	0.57
Roots	*Sarcocornia quinqueflora*	38.8	±	0.99
Roots	*Maireana oppositifolia*	44.5	±	0.22
Necro-mass				
Downed wood	*Avicennia marina*	47.8	±	0.35
Litter	*Avicennia marina*	41.2	±	1.45
Litter	*Sarcocornia quinqueflora*	36.7	±	3.50
Litter	*Maireana oppositifolia*	43.4	±	0.84

Separate analyses of the AG and BG carbon pools showed the same pattern as described for biomass. That is, the AG carbon pools (Table 7) were significantly different among all pair-wise comparisons ($F_{2,21} = 239.84$, $p < 0.001$; Tukey $p \leq 0.001$), with *A. marina* having the greatest AG carbon stock and *S. quinqueflora* the lowest. The BG carbon stock differed significantly among vegetation types ($F_{2,21} = 86.33$, $p < 0.001$); however, post-hoc tests showed that the BG carbon stock of *A. marina* and *S. quinqueflora* vegetation were similar (Tukey $p = 0.295$) and significantly greater than that of *M. oppositifolia* vegetation (Tukey p =<0.001; Figure 16) (see also Appendix A, Figure A.5, Table A.3).

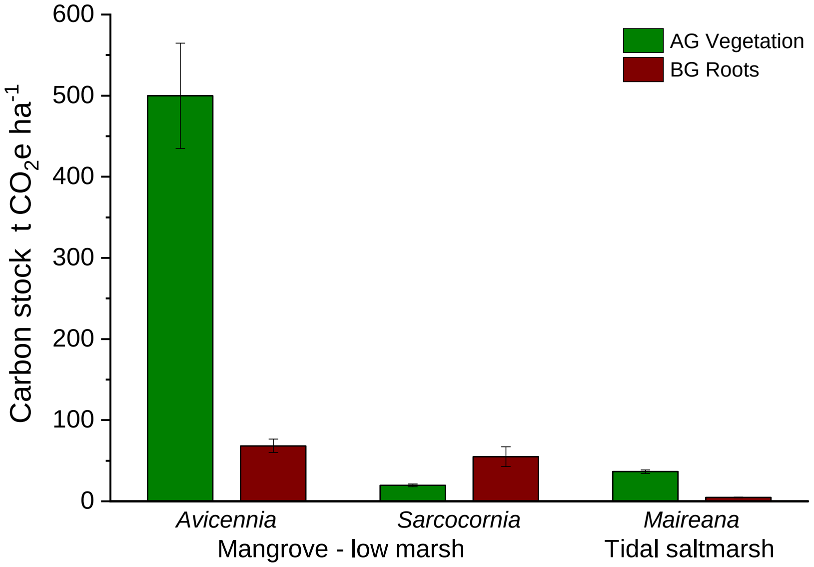

Figure 16. Carbon stocks (CO_2 equivalents; mean ± SE) of above-ground (AG) vegetation and belowground (BG) roots for three vegetation types: *Avicennia marina* forest, *Sarcocornia quinqueflora* saltmarsh and *Maireana oppositifolia* shrubland, corresponding with the strata 'Mangrove-low marsh' (for *Avicennia marina* and *Sarcocornia quinqueflora*) and 'Tidal saltmarsh'. AG = above ground, BG = below ground.

3.6 Summary and discussion of carbon dynamics following tidal reconnection

The carbon dynamics in soils and vegetation were assessed, along with GHG fluxes, during the early (ca. 1.5 years) stages of tidal reconnection and compared to reference areas. On average, there was a net gain of organic carbon stock in the soil following tidal reconnection. Visual observations and carbon and nitrogen isotope ratio measurements indicated this is mostly likely through seagrass wrack (allochtonous C) entering with the tide and being retained in the pond. Sediment accretion rates were highly variable and varied by several (± 3) mm yr^{-1} since tidal reconnection, and comparable to longer term sedimentation rates measured by ^{210}Pb dating and literature records for tidal wetlands (Villa and Bernal 2018). Methane gas fluxes were negligible, likely due to the high salinity in the pond.

The accumulation rates for organic carbon from the 'Mangrove-low marsh' stratum in the reference area were comparable to median values for mangrove and coastal wetlands obtained from a recent global synthesis of carbon accumulation rates (Wilkinson et al. 2018). The carbon accumulation rate in the trial pond was already initially high, possibly similar to patterns seen for impoundments (Wilkinson et al. 2018), and increased after tidal reconnection. This development corresponds to the global pattern emerging from the review by Wilkinson et al. (2018), which revealed nearly 4x higher carbon accumulation rates in tidal ecosystems compared to permanently inundated wetlands.

The measured values for carbon fraction of biomass of saltmarsh vegetation were similar to default conversion factors used in the blue carbon literature (e.g. Howard et al. 2014). The greatest carbon capture will be in mangrove owing to large above-ground biomass values. Roots were found to contribute a small proportion of mangrove biomass and, unexpectedly, mangrove root biomass was similar to that of *S. quinqueflora*. Mangrove root biomass could have been underestimated as small diameter cores tend to exclude larger roots and result in lower biomass estimates compared with more intensive sampling methods (e.g. trenches) (Adame et al 2017). Our data also indicated that saltmarsh can be a further important contributor to carbon sequestration. The AGB of *S. quinqueflora* was found to be approximately two times greater than a previous measure (Owers et al. 2018). Furthermore, our root samples indicate that the below ground C stock of *S. quinqueflora* is at least 2.8 times that of the above ground vegetation.

A summary of the carbon stock data for the carbon pool of vegetation and soil for each of the three strata is given in Table 10. The summary data are based on samples per stratum from soil carbon stocks (to 30 cm depths) and long term sedimentation, and for biomass also with consideration of the vegetation composition in each of the strata as informed by data from chapter 4. Saltmarsh vegetation rapidly colonised the pond following tidal restoration. Based on the elevation and predicted vegetation changes, over a longer period most of the pond is expected to be recolonised by mangroves. The feasibility of tidal reconnection for carbon sequestration has been demonstrated, and further calculations on long term carbon benefits under various scenarios are made in chapter 6 and the proof of concept report (Dittmann et al. 2019).

Table 10: Summary of carbon stock data of the main carbon pools based on strata specific samples from all project parts under the carbon dynamic task.

		Carbon stock					
		t C ha^{-1}			t CO_2e ha^{-1}		
STRATA	CARBON POOL	MEAN	±	SD		±	SD
Mangrove-low marsh	Biomass	175	±	67.3	643	±	246.8
	Soil	93	±	42.7	341	±	156.6
	Sum	274	±	122	1006	±	447.3
Tidal saltmarsh	Biomass	32	±	12.7	116	±	46.6
	Soil	127	±	90.2	465	±	330.7
	Sum	158	±	103	581	±	377.7
Supra-tidal saltmarsh	Biomass	32	±	12.7	116	±	46.6
	Soil	36	±	1.3	132	±	4.8
	Sum	68	±	14	248	±	51.3

4 Revegetation of trial salt pond

To determine the revegetation potential of the reconnected salt pond, several field investigations and experiments were carried out. The tidal wetland vegetation in the reference areas was characterised through transects. Changes in vegetation ground cover over time were assessed inside the trial pond and reference areas. The feasibility of recolonisation after reconnection was investigated with experimental assessments of seed dispersal and seed bank germination potential.

4.1 Vegetation surveys inside and outside of pond

The natural saltmarsh vegetation in the vicinity of the trial pond was surveyed to obtain an understanding of the potential plant species composition inside the pond following tidal reconnection. Three vegetation transects with elevation mapping were carried out in December 2017, from the top of the levy bank to the edge of the mangrove forest. Transects did not extend into the mangrove forest, which is composed of one species only. In the reference area, the transect had a length of about 400 m, while two transects of 200 m lengths were surveyed in the adjacent reference area due to the shorter distance between the levy bank and mangrove forest (Appendix B, Figures B.1 and B.3). Vegetation boundaries were recorded and quadrats placed along the transects surveyed using Biological Survey of SA methodology (Heard and Channon 1997).

The transects covered mostly the elevation ranges of the 'Tidal saltmarsh' and 'Supra-tidal saltmarsh' strata (Figure 17). Small-scale topography variation was pronounced at both sites, accompanied by small-scale variability in the presence of vegetation types (Appendix B, Figures B.2 and B.4). Vegetation types 6 (mostly dune species) and 1 occurred on top or along elevated cheniers. *Maireana oppositifolia* formed low closed shrublands on elevated substrate in vegetation type 5. *Tecticornia arbuscula* and *Sarcocornia quinqueflora* formed dense cover in vegetation types 3 (primarily *T. arbuscula*) and 4 (primarily *S. quinqueflora*). Vegetation type 7 was dominated by S. *quinqueflora* with juvenile mangroves and patches of *T. arbuscula*. Adjacent to the trial pond, *Suaeda australis* was prominent and formed a distinct vegetation type 8. The occurrence of saltmarsh plant species in this survey (Table 11) corresponded with elevation and tidal inundation records from Barker Inlet and Torrens Island (Fotheringham 1994; Fotheringham and Coleman 2008).

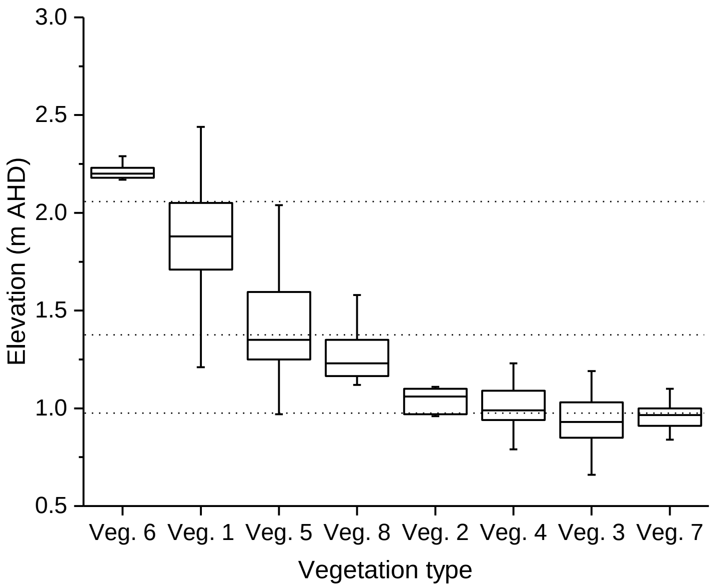

Figure 17. Box plots of elevation ranges across which the main vegetation types were recorded. The horizontal dotted lines indicate the upper and lower boundaries of strata (see Table 2.3.1). Vegetation types are described in Table 11.

Table 11: Species of the main vegetation types recorded in quadrats along the vegetation transects, with √ denoting abundances from covering <5% to >75% of the area, and o denoting sparse or very few (<10 individuals). * indicates introduced species. The sequence of vegetation types aligns with their occurrence along an elevation gradient (see Figure 17).

	VEG.6	VEG.1	VEG.5	VEG.8	VEG.2	VEG.4	VEG.3	VEG.7
Alyxia buxifolia	√							
Atriplex paludosa	√	√						
Olearia axillaris	√							
Grass	√							
Dianella brevicaulis	o							
Glycine clandestina	o							
*Lycium ferocissimum**	o							
Myoporum insulare	o							
Rhagodia candolleana	o							
Threlkeldia diffusa	o							
*Limonium companyonis**	o	√	√					
Maireana oppositifolia	o	√	√					
*Parapholis incurva**		√						
Spergularia media		√						
Tecticornia pergranulata		√						
Tecticornia arbuscula		√	o			o	√	√
Tecticornia halocnemoides		o			√			
Disphyma crassifolium		o						
Enchylaena tomentosa		o						
Nitraria billardierei		o						
Frankenia pauciflora		o	o					
Sarcocornia quinqueflora		o	√			√	√	√
Wilsonia humilus			√					
Suaeda australis			o	√				
Sarcocornia blackiana			o					
Hemichroa pentandra			o			o		
Samolus repens			o			o		
Avicennia marina								√

4.2 Seed dynamics experiments

The revegetation of saltmarsh and mangrove is subject to seed or propagule availability from the local seed bank, dispersal, and suitable hydro- and morpho-dynamic conditions (Friess et al. 2012; Rand, 2000; Zhu et al. 2014). To understand the revegetation pathway of a reconnected salt pond, field experiments were carried out in the trial pond and reference areas. The objective was to compare seed dispersal and soil seedbanks at established saltmarsh sites with that of the trial pond in order to gain insights into the early colonisation of plants facilitated by tidal reconnection. We expected that a greater number of seeds would occur at the established marsh sites compared with the trial pond and that the species composition within the pond would be more similar to the adjacent reference area than the reference area (ca. 1 km away). As seed delivery and establishment can be affected by a window of opportunity with suitable hydrological and geomorphic conditions (Balke et al. 2014; Spencer and Harvey 2012), we also determined physical factors and elevation that may influence seed rain and soil seedbanks within the trial pond.

4.2.1 METHODS

Seed dynamics were investigated inside the trial pond (TP) and at a relatively intact saltmarsh and mangrove area at the reference area (RA). The adjacent reference area (AR) next to the pond inlet was selected for its potential as a seed source for revegetation of the trial pond, given its proximity to the pond inlet. Sample locations covered the strata for 'Mangrove-low marsh' and 'Tidal saltmarsh', but are differentiated here as 'low' and 'mid' elevations, as no accurate bathymetry was available at the time of first trap deployment for all locations inside the pond and reference areas. Sample size for each of the mid and low elevations was n = 8 per elevation at the reference, and n = 18 per elevation in the trial pond sites. Low elevations were similar at the two sites, although locations at mid elevations were relatively higher at the reference area compared to the pond (Appendix B, Figures B.5), largely due to abrupt shifts to higher elevations as a result of shell grit cheniers present in the reference area but not discernible within the trial pond. Due to the pattern of elevation within the pond, all traps close to the main channel were at slightly higher elevations than those placed away from the main creek. Sample locations at the adjacent site (n = 6) were not stratified by elevation and ranged from 0.80–1.11 m AHD.

Seed traps

For seed traps, astroturf® mats (30×30 cm) were used to assess seedfall as a result of hydrochory (dispersal by water) (Wolters et al. 2004). Within the pond, a total of 36 traps were positioned on each side of the main creek, either near to (2.8 $\pm$ 1.01 m from the creek bank) or far from the creek (41.3 $\pm$ 7.81 m from the bank). Due to accessibility within the pond, most traps were placed within 260 m of the pond inlet. Four traps were placed near the furthest extent of the main creek (ca. 530 m from the pond inlet).

In the adjacent reference area, six traps were placed within 60 m of the pond inlet to assess a potential supply of dispersing seeds. At the reference area, seed traps were placed in mid- and low-marsh locations, distributed across the accessible extent of the mature saltmarsh, but irrespective of the distance to the creek due to the terrain at the site. Where possible, traps were set up within small vegetation gaps, although any vegetation within 20 cm of the traps was clipped to ensure trapped seeds were unlikely a result of primary seed dispersal.

Seed traps were first set in December 2017 and then collected and replaced at approximately one month intervals for a period of 12 months. To remove the seeds and other material, traps were hung within a tall bucket and thoroughly rinsed with pressured water. A small number of traps were covered with wrack on occasions, which was collected with the seed traps and rinsed over a 4 mm sieve. The resulting samples from traps (and associated wrack) were passed through a 0.25 mm sieve and the retained material spread over seedling trays (16×10×5.5 cm) filled with 400 mL of sand (including ca. 2.2 g of slow release fertiliser per tray).

Samples were grown in a glasshouse where the temperature was allowed to fluctuate freely between 16 and 26^{o}C. Samples were watered 2–3 times per 24 hours with freshwater and kept in these conditions for three months, by which time seedling emergence had ceased. During the growing period, identifiable seedlings were counted and removed while reference specimens of unidentifiable plants were placed in pots and grown to maturity. *Tecticornia* and *Sarcocornia* species can be difficult to distinguish as juveniles and thus, seeds were collected from plants in the field and grown in the glasshouse. These species were found to be most effectively distinguished based on morphology of the seedlings at the cotyledon stage.

Seed bank

Soil seed bank samples were collected in February, May, August and November of 2018. The sampling in February occurred before new plants within the trial pond produced seeds. For each trap location, two subsamples (bulked in the field) were collected at a distance of 3 m and in random directions from the traps. Each subsample was taken by pressing a 8×10.5 cm rectangular tube into the soil and excavating the soil to a depth of 3 cm. The samples were rinsed first through a 4 mm sieve to remove large debris and the material then retained in a 0.25 mm sieve (TerHeerdt et al. 1996) placed in seedling trays (33×13×5.5 cm) on a layer of sand (500 mL including ca. 2.2 g of slow release fertiliser per tray). Growing conditions and data collection were the same as for the seed trap samples (see above). This method provides an estimate of the germinable seeds in the seed traps and the soil seed bank samples.

Plant phenology

The reproductive phenology of common plant species was characterised near to the seed traps. Specifically, the occurrences of reproductive states were recorded monthly for each plant species located within two, 1 m^2 quadrats, which were positioned 0.5 m from either side of the traps. Reproductive states included; 'vegetative' (non-reproductive juvenile or mature plants), 'flowering', 'immature fruit' (green, fleshy fruits) and 'mature fruit' (dried fruit with evidence of shedding seeds). Reproductive phenologies of *Sarcocornia quinqueflora*, *Suaeda australis*, *Maireana oppositifolia* and *Wilsonia humilis* were assessed by determining, for each month, the number of quadrats in which a given species' reproductive state was observed, out of the number of quadrats in which the plant species occurred in. We were unable to characterise differences in immature and late fruit for *M. oppositifolia*. In addition, the number of dead and alive mangrove propagules within quadrats were recorded monthly.

Biophysical factors and seed availability

Several variables that may influence the spatial patterns of seed fall within the pond were measured. Elevation (m AHD) was measured at each trap location using a differential GPS. The weight loss of Plaster of Paris blocks was used to assess the relative inundation and erosive force (Thompson and Glenn 1994) of tidal flows at the seed trap location. Molded plaster blocks (21.9 $\pm$ 0.06 g, n = 58) were glued to the upper surface of plastic petri dish lids, which allowed the units to be pegged to the sediment surface through a hole drilled in the petri dish lids. Plaster blocks were deployed in May 2018 during the suspected main seed fall period and collected 33 days later. The units were dried in a 40^{o}C oven for 72 hours and weighed to calculate the weight lost from the plaster blocks. Finally, the proximity of traps near to the main channel was defined as ‘near’ and ‘far’ and the linear distance of each trap location from the pond inlet was calculated from a mapping system. In June 2018, a proxy of local seed availability was obtained by counting the number of reproductive plants (almost entirely *S. quinqueflora* and *S. australis*) within a 5 m radius of each seed trap within the pond.

Data analysis

Generalised linear models (GLMs) were used to test for differences in seed abundance between sites (pond and reference) and elevations (mid and low elevations). Abundances of germinable seeds for trap and soil seed bank samples were scaled to 1 m^2 and non-saltmarsh species, which accounted for less than <2.5% of the seeds, were excluded. Data for the adjacent site were not included in the analyses as samples were not stratified by elevation and variation in seed densities were found to be extreme. Due to over-dispersion, seed trap and soil seed bank data were modeled on quasi-Poisson distributions using a log-link function. Seed trap data were summed over sample months, providing estimates of the seed fall density over the 12-month period. Soil seed bank data were averaged (rather than summed) over the four sampling months (February, May, August and November). Averages were calculated due to the potential for seed bank samples to contain seeds that had accumulated over unknown periods of time. The analyses described were also conducted for *S. quinqueflora* only, given the seed pools were dominated by this species.

Temporal patterns of seed abundance were explored graphically. Seed trap data were pooled over three-month periods (i.e. January+February+March), with the ‘middle’ months of each time period coinciding with the soil seed bank samples.

Non-metric multidimensional scaling (nMDS) was used to visualise differences in the saltmarsh species composition within the soil seed bank and seed trap samples, based on data grouped over the sampling periods. Permutational multivariate analysis of variance was used to test for overall compositional differences between site/elevation combinations for seed trap and soil seed bank data.

Non-hierarchical linear regression was used to explore biophysical factors within the pond that may influence the density of seedfall of *S. quinqueflora* and *S. australis*. The abundances of the two species in seed traps were summed over the sampling months and log10 transformed. The explanatory variables included were: 1) trap elevation (m AHD); 2) the trap distance from the pond inlet; 3) the trap distances from the main channel (defined as a categorical variable, ‘near’ and ‘far’); and 4) the number of reproductive plants (either *S. quinqueflora*, or *S. australis*) within 5 m of the traps. The weight loss of plaster clods was strongly correlated with elevation (Pearsons correlation = -0.737, $p < 0.001$) and thus, this explanatory variable was excluded from the analyses.

Analyses were conducted in R (R Development Core Team 2017) using lme4 for GLMs (Bates et al. 2015) and vegan for nMDS (Oksanen et al. 2018).

4.2.2 RESULTS

Plant phenology

The fruit of *Maireana oppositifolia, Sarcocornia quinqueflora* and *Suaeda australis* mainly reached maturity and released seeds in the autumn months and this extended into June and July for the latter two species. *Wilsonia humilis* bore mature fruit from January to May (Figure 18). Mature fruit of *Sarcocornia quinqueflora* and *Suaeda australis* plants within the pond was first observed in April 2018. Monthly quadrat counts of mangrove propagules indicated that viable propagules disperse into the pond and are available from November to March (see Appendix B, Figure B.6). Densities of mangrove propagules within the trial pond peaked at 0.15 ± 0.066 viable propagules per m^2 in December 2018.

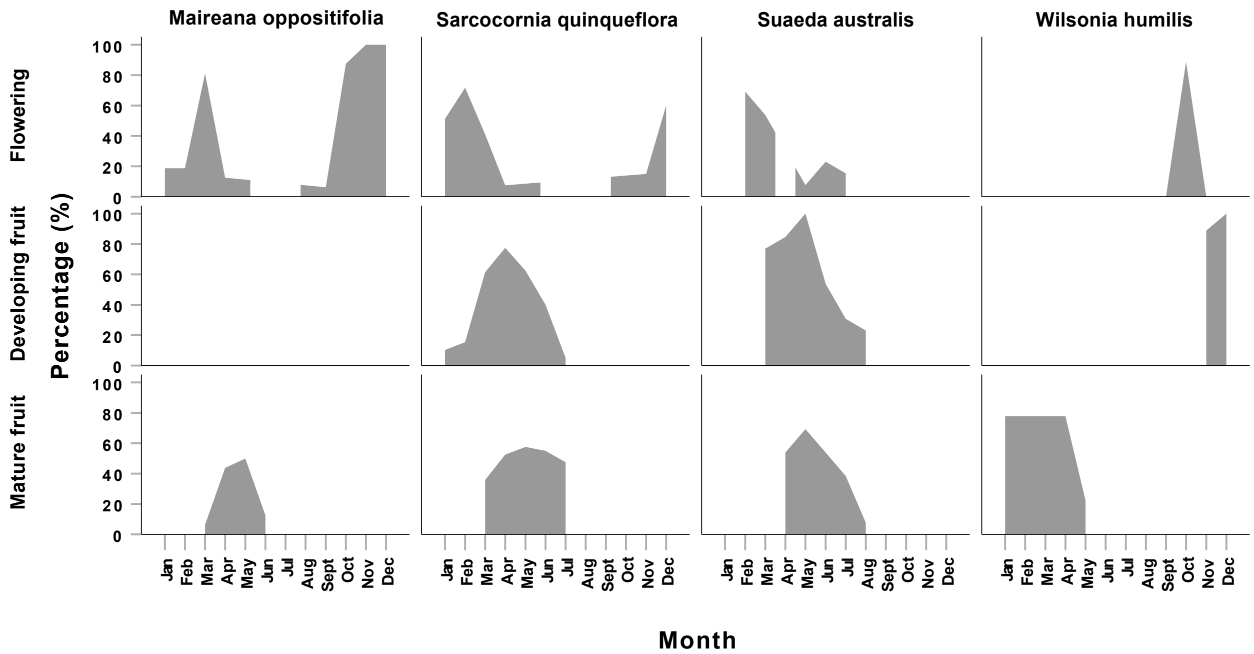

Figure 18. Reproductive phenology of common saltmarsh plants over the course of a year (2018). For a given plant species, percentages are based on the number of quadrats a reproductive stage was evident out of the number of quadrats that the given species occupied (*S. quinqueflora*, n = 39-40; *S. australis*, n = 13; *M. oppositifolia*, n = 16; *W. humilis*, n = 9). More than one reproductive stage may be recorded for a given quadrat. The distinction between developing and mature fruit of *M. oppositifolia* was difficult to determine, and thus, the presence of fruit is displayed as mature fruit.

Seed dispersal and soil seed banks

A total of 33 species were detected in the seed traps and/or the soil seedbank samples, 13 of which were deemed saltmarsh plants (Appendix B, Table B.1). Overall, *S. quinqueflora* comprised 74% of the plants in both traps and soil seedbanks, while *S. australis* accounted for 16.6% and 18.4% of the plants in traps and seedbank samples, respectively. Twenty species of non-saltmarsh plants were uncommon and together accounted for <2.5% of germinants from both traps and seedbank samples.

The highest mean density of saltmarsh seeds per trap location (summed over 12 monthly collections) and per seed bank sample (averaged over four sampling occasions) occurred at the adjacent reference area. Densities were 4946 $\pm$ 1624.6 and 2827.4 $\pm$ 936.7 seeds per m^2 for seed trap and soil seed bank samples, respectively (Appendix B, Table B.1). Seeds mainly consisted of *S. quinqueflora* and *S. australis*.

Analysis of the data from the pond and reference sites showed that the density of saltmarsh seeds within seed traps depended on a significant interaction between site (Pond and Reference site) and elevation ($t = -2.79$, $p = 0.0075$). Densities were higher at mid elevations compared with lower elevations but the magnitude of this difference was greater for the pond site compared with the reference site (Figure 19). Across the two sites, the density of seeds was highest at mid elevations within the pond (along the banks of the main channel) and lowest at low elevations within the pond (1772.8 $\pm$ 350.8 vs 111.1 $\pm$ 14.9 seeds per m^2; Appendix B, Table B.1). These patterns were largely due to the predominance of *S. quinqueflora*, the density of which also depended on a significant interaction between site and elevation ($t = -2.78$, $p = 0.0078$, Figure 20).

Seed bank densities also depended on a significant interaction between site and elevation strata ($t = -2.78$, $p = 0.0078$), but the pattern of seed density differed from that of seed traps (Figure 19). More seeds occurred at mid elevations within the pond than at lower elevations (169.5 $\pm$ 38.0 vs 16.5 $\pm$ 4.0), while the opposite was found at the reference site (303.2 ± 58.0 vs 770.1 ± 131.0, Figure 19). Again, the pattern described was largely driven by the high densities of *S. quinqueflora*, with a significant interaction between site and elevation ($t = -4.43$, $p < 0.001$, Figure 20).

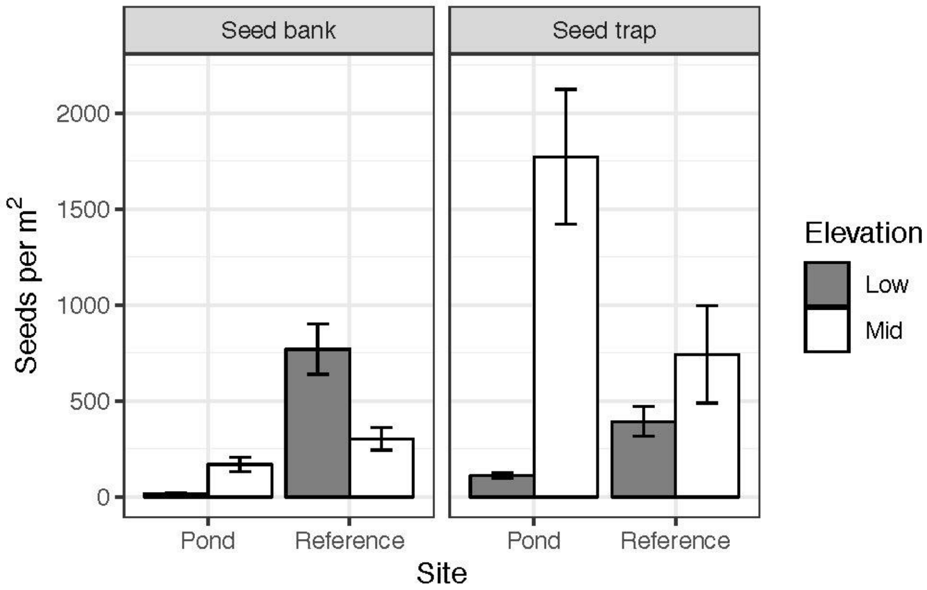

Figure 19. Mean ± SE number of seeds per m² for seed bank and seed trap samples collected from the trial pond and the reference site. Seed bank samples were taken in February, May, August and November 2018 and seed densities are averaged over the sample months. Seed trap samples were taken ca. monthly and seed densities are summed over the 12 months.

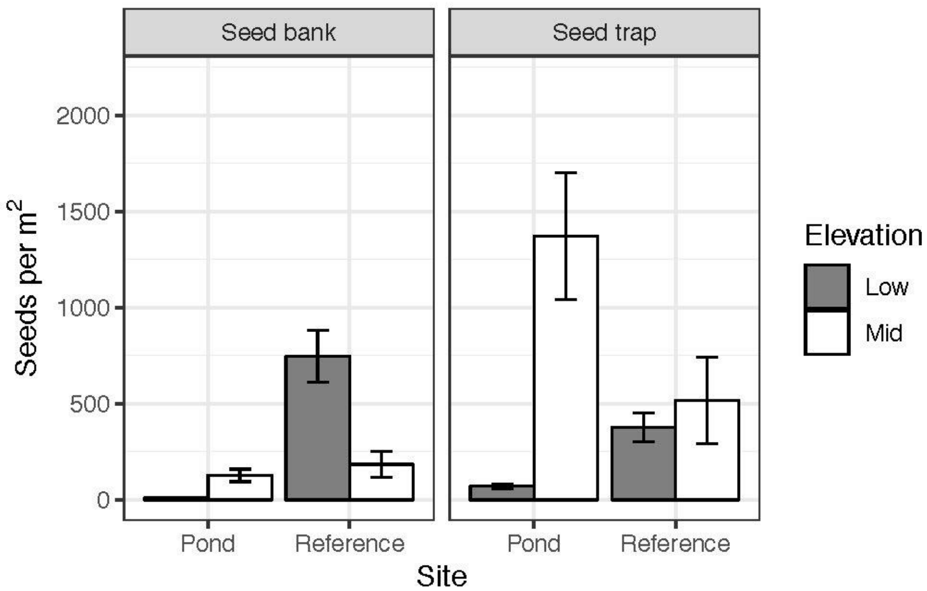

Figure 20. Mean ± SE number of *S. quinqueflora* seeds per m² for seed bank and seed trap samples collected from the trial pond and the reference site. Seed bank samples were taken in February, May, August and November 2018 and seed densities are averaged over the sample months. Seed trap samples were taken ca. monthly and seed densities are summed over the 12 months.

Seedfall showed a distinct peak at all sites during the April, May and June month grouping and monthly trap data showed that this peak was largely restricted to the traps collected in mid-June (Figure 21). Negligible amounts of seed fall occurred in January, February and March 2018, a period where the first recruits within the trail pond were yet to produce seeds. The density of seeds in the soil seed bank samples also showed a temporal pattern. Within the pond, densities were greatest at mid elevations for the August 2018 sample. At the reference site, densities peaked at mid elevations in the May sample, while at lower elevations, higher densities of seeds persisted for the May, August and November samples (Figure 22).

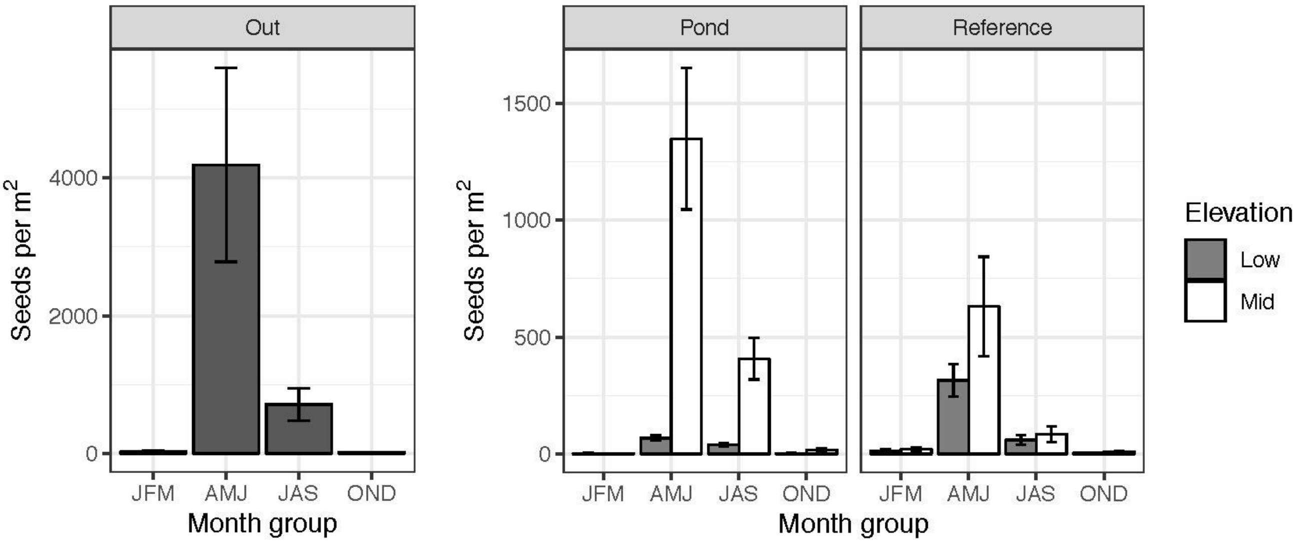

Figure 21. Mean±SE number of seeds per m^2 collected from seed traps at the adjacent reference area, the trial pond and reference site. Data from monthly seed trap samples are summed over three-month periods (e.g. JFM; January, February, March).

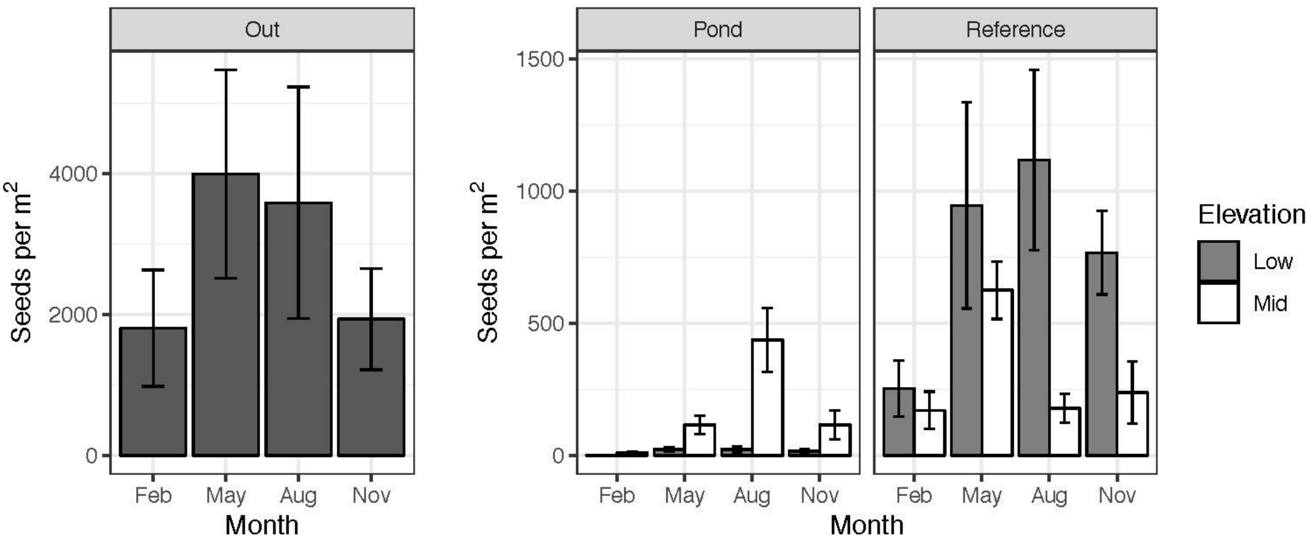

Figure 22. Mean±SE number per m^2 in the soil seed bank in the adjacent reference area (Out), the trial pond and reference site.

Site/elevation combinations significantly affected the community composition of seed fall (pseudo-$F_{4,53}$ = 10.49, R^2 = 0.44, P <0.001) and seed banks (pseudo-$F_{4,44}$ = 7.38, R^2 = 0.40, P <0.001). Based on the ordination plots, distinct differences in community composition between the reference and pond sites were evident for both the soil seed bank and seed trap samples (Figure 23). Furthermore, compositional differences were evident between both elevation strata at the reference site. The composition of the adjacent reference area was similar to the pond due to the dominant combination of *S. quinqueflora* and *S. australis* at both sites.

Biophysical factors and seed dispersal

The regression model for *S. quinqueflora* ($F_{4,31}$ = 29.62, R^2 = 0.7659, p <0.001) indicates that the density of seed fall within the pond was significantly influenced by the distance from the pond inlet (β = 0.001, $t_{4,31}$ = -2.304, p <0.028), the number of reproductive plants within the vicinity of seed traps (β = 0.029, $t_{4,31}$ = 4.291, p <0.001), and the distance to the main channel (β = 0.594, $t_{4,31}$ = 3.343, p = 0.002). The model for *S. australis* ($F_{4,31}$ = 5.74, R^2 = 0.3517, p = 0.001) indicates that the main variable influencing the density of seeds is the number of reproductive plants (β = 0.189, $t_{4,31}$ = 3.622, p = 0.001), while inlet distance, channel distance and elevation were not significant predictors.

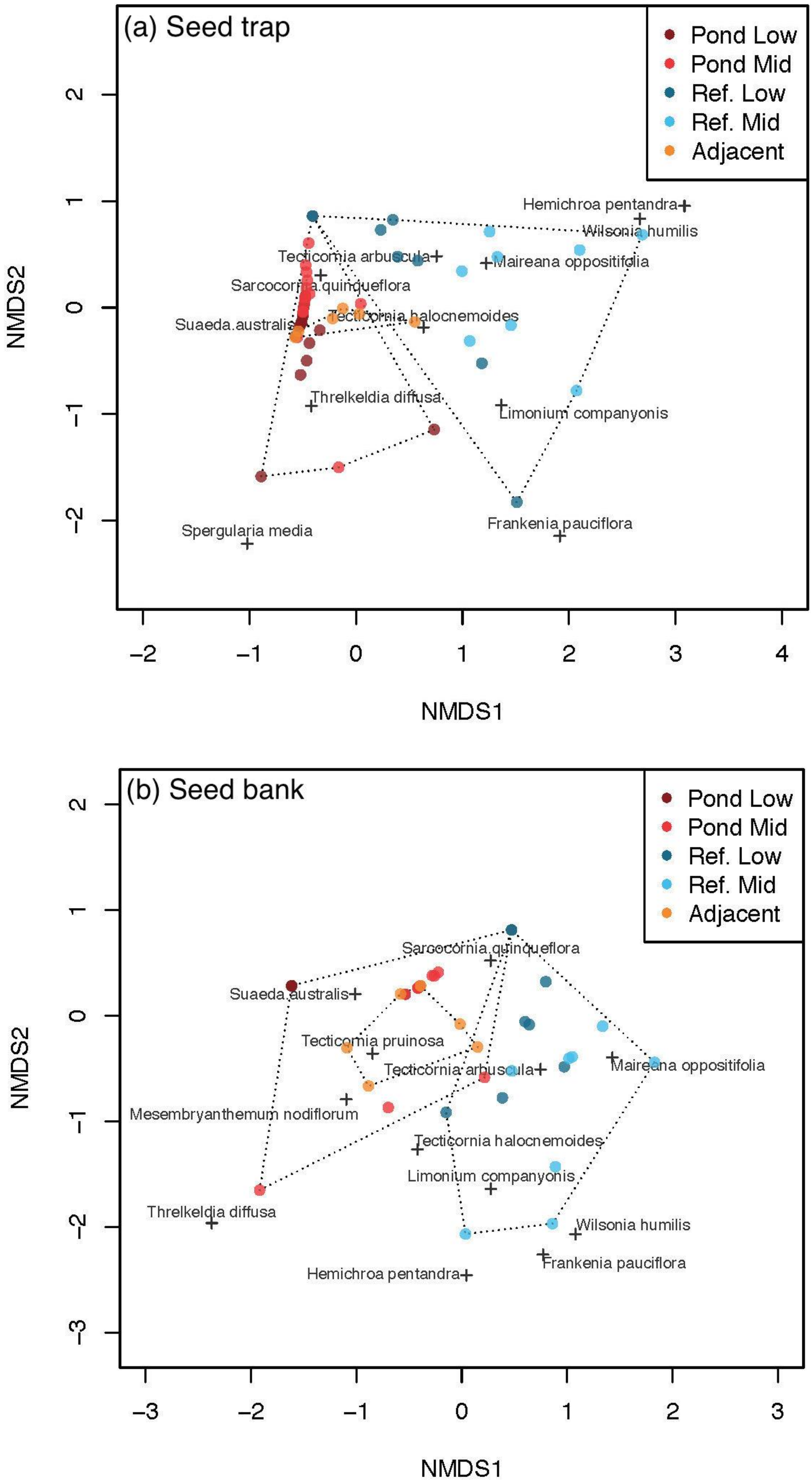

Figure 23. Non-metric MDS plots of saltmarsh species composition for the soil seed bank (stress = 14.4%) and seed trap (stress = 11.9%) samples. Seed trap data are based on the number of seeds per m^2, pooled over 12 monthly samples. Seed bank data are based on the number of seeds per m^2, averaged over four sampling occasions. Note, some points overlap and samples with no saltmarsh species cannot be included in the ordinations.

4.3 Revegetation inside trial pond and plant cover

4.3.1 METHODS

Following reconnection in July 2017, the first seedlings of two chenopod species, *Sarcocornia quinqueflora* and *Suaeda australis,* were observed along the banks of the main creek inside the trial pond in mid-November 2017. To further document the progress of commencing natural revegetation within the trial pond, ground cover was measured in Autumn, Winter, Spring and Summer of 2018, using the point-intercept method. Point intercepts were recorded along 2 x 5 m transects associated with each seed trap (section 4.2). Transects originated at 0.5 m from the centre of each seed trap to avoid areas where vegetation may have been removed or affected by trampling. Transects for ground cover vegetation were carried out in the trial pond (TP), adjacent to the pond (AR) and the reference areas (RA). The transects inside the trial pond ran parallel to the main creek (approx. north-south direction), while all other transects extended in a north-south direction.

The cover of plant species and other ground-cover components (logs, seagrass wrack and bare ground) that contacted a vertical pole were recorded at 10 cm intervals along each transect. Logs greater than ca. 2 cm in diameter were only recorded if contacts with the vertical pole were made <0.5 m above the ground. Bare ground was only recorded where its presence was not associated with any other ground cover type. Percentage ground cover at each trap location was calculated as the number of 'hits' out of 102 points per trap location; i.e. point intercept data were pooled over the two 5 m transect associated with each seed trap.

Differences in the composition of ground cover between site/elevation combinations and season were characterised using non-metric multidimensional scaling (nMDS) based on the Bray-Curtis dissimilarity index. The ordination was performed on ground-cover data from all four sample seasons. Due to the absence of vegetation within the pond at many sampling points, non-living ground cover elements (dead wood, seagrass wrack and bare ground) were included in the ordination. Compositional differences between site/elevation combinations and season were tested using a permutational multivariate analysis of variance (Oksanen et al. 2018).

4.3.2 RESULTS

A total of ten saltmarsh plant species were observed in the ground cover. At the reference area, plant cover remained stable over the seasons, ranging between 88.1 $\pm$ 7.20% (low elevation, Spring) and 96.4 $\pm$ 1.53% (mid elevation, Spring). Plant cover at the adjacent area ranged between 65.8 $\pm$ 8.45% and 76.6 $\pm$ 8.78%. Within the pond at mid elevations (along the banks of the main channel) plant cover progressively increased from 3.87 $\pm$ 1.23% in Autumn 2018 to 40.25 $\pm$ 5.24% in Summer 2018. At low elevations away from the main channel, plant cover reached 3.27 $\pm$ 1.22% in Summer 2018 (Figure 24a). *Suaeda australis*, *Sarcocornia quinqueflora* and *Threlkeldia diffusa* were the only species detected within the pond by the ground cover surveys and the former two species listed were dominant (Figure 24b, c).

The composition of ground cover was significantly different between site/elevation combinations (pseudo-$F_{4,212}$ = 136.32, R^2 = 0.69, P <0.001) and season (pseudo-$F_{3,212}$ = 3.82, R^2 = 0.01, P <0.001). Clear compositional differences were evident between the reference site, mid and low marsh areas. The ground cover composition of the adjacent reference area was distinct from the reference area, in part due to the dominance of *S. australis* at the adjacent area outside the trial pond. The ground cover composition of the pond remained distinct from all other sites, with wrack, downed wood (mainly dead mangroves) and bare ground being major features (Figure 25a). There was a marginally non-significant interaction effect between site/elevations and season (pseudo-$F_{12,212}$ = 1.411, R^2 = 0.02, P = 0.058), which is likely due to the directional shift in the pond groundcover composition towards that of the adjacent site. From autumn 2018 to summer 2019, the composition of ground cover at mid-elevations within the pond becomes more similar to the adjacent site, largely due to the colonisation of the pond by *S. australis* and *S. quinqueflora* (Figure 25b). The composition at low elevations of the pond also shifted in the direction of the reference site.

Propagules of mangrove were recorded along the creek banks inside the pond within six months after tidal reconnection, and several appeared to grow roots. Mangrove propagules readily dispersed into the pond, mainly during the summer months (Appendix B, Figure B.6). Propagules were observed at the furthest extent of the main channel, but large quantities collected closer to the pond inlet and on the downslope of the main channel, in particular. Several seedlings were observed to take root, but by the end of the study period had not established successfully.

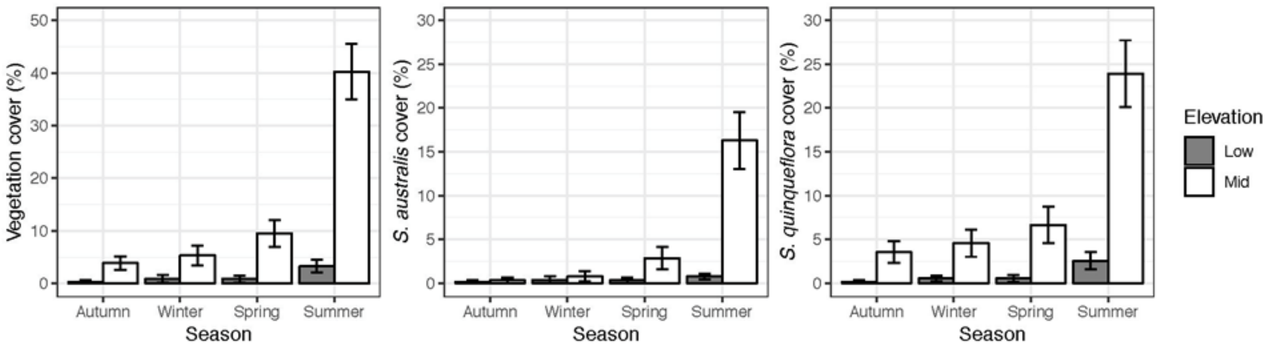

Figure 24. Mean±SE plant cover (%) of line intercept transects within the trial pond recorded in March (Autumn), June (Winter), September (Spring) and December (Summer), 2018. Panels show (a) total plant cover and cover for the two dominant species; (b) *Suaeda australis* and (c) *Sarcocornia quinqueflora*. Within the pond, mid elevations (white bars) occur along the banks of the main channel while low elevations (grey bars) occur away from the main channel.

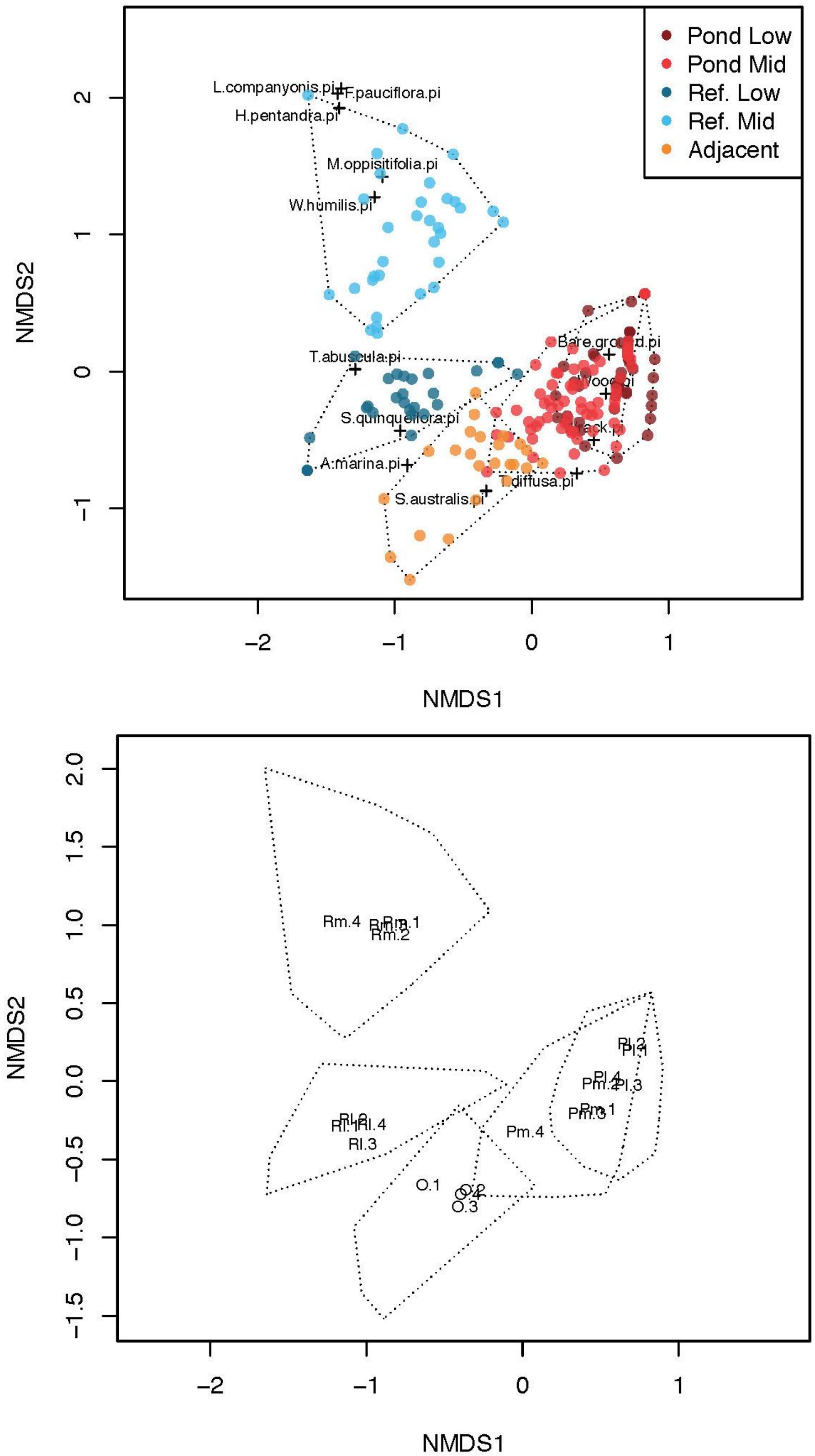

Figure 25. Non-metric MDS plots of ground cover (plant species, wood, wrack and bare ground) at mid and low elevation strata of the reference area and the trial pond, and adjacent reference area to the pond inlet. Dotted polygons encircle strata. a) Samples (seed trap locations) and ground cover types. b) Centroids for each strata-by-season combination. Centroid codes are as follow; P = Pond, O = Adjacent, R = Reference; m = mid elevation, l = low elevation; 1, 2, 3 and 4 represent Autumn, Winter, Spring and Summer samples, respectively. Stress value = 12.96%.

4.4 Summary and discussion of revegetation investigations

The revegetation experiments have documented a fast recolonisation by saltmarsh plants, dominated by pioneer species *Suaeda australis* and *Sarcocornia quinqueflora* which grew quickly to mature stages inside the trial pond. Seed fall, soil seed banks and vegetation cover were assessed at an established saltmarsh (reference site), the trial pond and at a site adjacent to the pond inlet. The findings of this study provide a unique insight into the early stages of revegetation of a salt pond following tidal reconnection and are informative for projecting large-scale restoration opportunities.

Restoration efforts of tidal wetlands often rely on adjacent saltmarsh as a natural source of seeds. A supply of seeds is particularly important for restoration of agricultural land and salt ponds where plants and propagules have been extirpated as a result of the land conversion (Bakker et al. 2002, Diggory and Parker 2011). In the current study, recruitment of *S. quinqueflora* and *S. australis* within the trial pond was first observed in November 2017, reflecting previous observations that these species rapidly colonise newly created saltmarsh sites (Fotheringham 2016). Following tidal reconnection in July 2017, seeds of *S. quinqueflora* and *S. australis* could enter the trial pond by tidal waters (hydrochory) from the adjacent site, where they dominated the vegetation. The magnitude of this first seed influx was not recorded, although very low numbers of *S. quinqueflora* and *S. australis* seeds were detected in the January–March seed trap samples inside the trial pond.

Measurements of vegetation cover, seedfall and soil seed banks all suggest that the short-term development of vegetation within the pond was strongly influenced by the vegetation immediately adjacent to the pond inlet. The composition of the seedfall and soil seed banks within the pond was more similar to the adjacent site than to low and mid elevations of the reference site. Furthermore, the ground cover composition shifted towards that of the adjacent site over time as the cover of *S. quinqueflora* and *S. australis* increased. *S. australis* accounted for ~27% of the seed bank and seed trap samples at the adjacent site, which was comparable to that found along the banks of the main channel within the pond (24.1% and 26.1% of saltmarsh seeds). *Suaeda australis* occurred as seeds in trace amounts at the reference site and was not detected in the standing vegetation.

While the establishment of saltmarsh inside the reconnected trial pond was initially reliant on influx of seed from adjacent areas, seed production by the first cohort of plants inside the pond provided a local source of seeds after ca. one year of tidal reconnection. Mature fruits were observed in autumn and winter 2018, and this was followed by a large increase in the number of dispersing seeds and the size of the soil seed bank along the banks of the main channel. Furthermore, a second recruitment event was initiated in Jul 2018, which contributed to the rapid increase in plant cover from spring to summer 2018. The results are evidence that fast recolonisation of salt ponds is possible once reconnected, although limited seed dispersal and consequently recruitment occurred at lower elevations away from the main channel, at least within the short duration of this study. Timing the reconnection with the optimum season for seed dispersal, as happened in this study, can accelerate the revegetation.

Tecticornia arbuscula is an important component of low marsh areas in south-eastern Australia (Saintilan 2009), and was co-dominant with *S. quinqueflora* at the reference site and thus is a defining feature of the species composition. Even so, this species accounted for only a small fraction (1.0–3.2%) of saltmarsh seeds in trap and soil seed bank samples at the reference site. Low numbers of seeds were also detected at the adjacent site and only 3 seeds were detected in the pond. This indicates a very low capacity for rapid colonisation by *T. arbuscula*. Indeed, *T. arbuscula* is considered to have limited dispersal ability and low seed production relative to other samphire species (Coleman et al. 2017) and was largely absent from sites on Torrens Island, two years after restoration of a sand mine site (Fotheringham 2016).

For the pioneer saltmarsh plants, the tidal inundation and disturbance free period would have provided a window of opportunity which favoured their fast re-establishment (Balke et al. 2014). Ongoing biogeochemical interactions with establishing vegetation can have positive effects on sediment surface elevation (Friess et al. 2012; Silinski et al. 2016), which can enhance the carbon sequestration rates in reconnected ponds. Our study covers just one year of revegetation, and indicates an early trajectory of increased carbon sequestration.

Following principles of calculations for carbon stock changes in shrubs (section 10 in AR-Tool 14), the measured carbon content for above and below ground biomass for *S. quinqueflora* per ha (see chapter 3, Table 10), and the increase in per cent cover from transects at low and mid elevations (Figure 24), an increase in carbon stock within the trial pond was estimated to be 122 ± 23.04 t CO_2e in a year (2018). The estimate also used the known elevation range for *S. quinqueflora* (Fotheringham 1994) and the extent of this elevation inside the trial pond. Following the first appearance of saltmarsh seedlings inside the trial pond in late November 2017, this increase in carbon stock was most pronounced at the mid elevation (Figure 26). While this calculation could be an over-estimate as less extensive below-ground biomass would be established during early colonisation, the value was for *S. quinqueflora* only. Further studies should also obtain biomass and carbon data for *Suaeda australis*, which was one of the first colonisers and accounted for nearly half of the emerging saltmarsh vegetation inside the trial pond (Figure 24). The estimate thus gives an indication of the change in carbon stock for saltmarsh vegetation following reconnection of the salt pond.

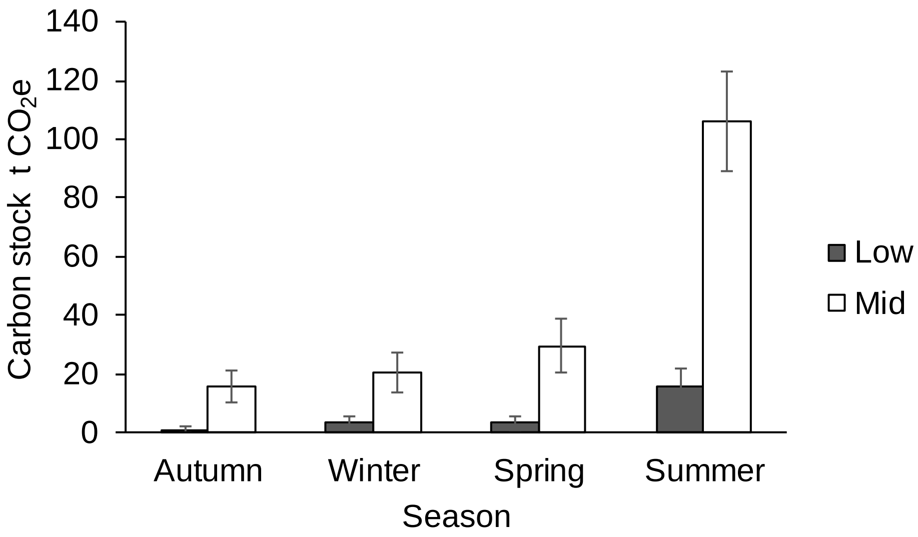

Figure 26. Estimated change in carbon stocks (mean ± SE) for the increase in *Sarcocornia quinqueflora* vegetation within the project boundary of the trial pond during 2018. The calculation is based on the elevation range for the species, the area suitable for colonisation in the pond, the measured rate of increase of plant cover, and the carbon content determined in task 1.

Within the first year of re-colonisation, the succession was still at a pioneer stage and characterised by few species in increasingly high abundance. Mangrove propagules were noted inside the pond in the first months after tidal reconnection, but did not establish during the study period. The adjacent mangrove forest provides an important source for propagules, as long distance dispersal of *Avicennia marina* propagules is less common (Clarke 1993). The window of opportunity for establishment of *Avicennia alba* includes thresholds linked to an inundation free period to develop roots, and development of long enough roots which allow the seedling to withstand hydrodynamic forces and dislodgement (Balke et al. 2011). During the early period after reconnection, a similar window of opportunity for establishment of *A. marina* inside the pond may not have been met. Improvement to sedimentary conditions and shelter among the now established pioneer saltmarsh could facilitate establishment of *A. marina* seedlings in coming seasons.

5 Carbon accounting and registration

To enable a blue carbon project to generate carbon credits, mechanisms have to be in place, and approaches followed as per respective accounting methodology. This project has become influential for developing a respective method for the ERF based on the international VCS method (see chapter 7). Here, we present a brief overview on the carbon offset opportunities in the Australian context, the international voluntary market, and evaluate whether the 'Human-Induced Regeneration' method under the ERF could be an alternative method to use. We also present an assessment of eligibility of tidal reconnection under criteria for Australia's offset integrity standard in an approach similar to a previously applied evaluation of blue carbon project activities (Kelleway et al. 2017a). A proof of concept for tidal reconnection, following a Verified Carbon Standard (VCS) methodology approach, is compiled in a separate report (Dittmann et al. 2019). Projected carbon abatement values within three decades are also presented in chapter 6 for the trial pond and further scenarios for the Dry Creek salt field.

5.1 Blue carbon offset in the Australian and international context

As Party to the United Nations Framework Convention on Climate Change (UNFCCC), Australia has committed to mitigate climate change through emissions reductions and Nationally Determined Contributions, with progress tracked through National Greenhouse Gas Inventories (NGGI) (Bell-James 2016; Herr et al. 2019). The NGGI align with the main categories set by IPCC Guidelines for good practice (IPCC 2006, IPCC 2014 (2013 Wetland Supplement)), and included mangrove and saltmarsh (as loss from extraction activity) since the 2015 NGGI report. In Australia, the Commonwealth Government is assessing the inclusion of additional blue carbon sinks and sources in the NGGI.

Carbon credits can be gained in Australia under the ERF, enabled through the *Carbon Credits (Carbon Farming Initiative) Act 2011* (Bell-James 2016). This can occur through a 'reverse auction' process enabling contracts with the CER or through separate registration under the ERF. At the last auction by the CER in December 2018, the price per Australia carbon credit units (ACCUs) was $13.87. Trading in ACCUs on-the-spot market in May 2019 has seen the price rise to $16.50. Projects can only be registered with the CER when a respective method exists in the ERF, which is not yet the case for blue carbon. The development of such a method has been explored (Cannard et al. 2016; Kelleway et al. 2017a), and is currently gaining momentum. Further steps in the development of blue carbon method under the ERF include a feasibility assessment of project activities, such as the introduction of tidal flow, and evaluation of the suitability of the VCS method VM0033 for wetland restoration (see also chapter 7).

The CER sets several criteria for offsets integrity standards, which must be met by a methodology to be eligible under the ERF. These criteria include additionality (the abatement is unlikely to occur in the ordinary course of events), whether the emissions reduction can be accurately measured and verified and be supported by evidence, as well as eligibility considerations for inclusion in Australia's GHG inventory and international reporting obligations. Other aspects to consider include legal or property rights, and leakage (increase in emissions outside of the project area). The timeframe of projects has to be defined as well.

Carbon offset and emissions trading are theoretically possible through international agreements as well as voluntary markets (Crooks et al. 2019; Needelman et al. 2018, 2019). All carbon offset markets use mechanisms or standards for a verifiable and accountable approach to issue carbon credits. The Clean Development Mechanism (CDM) was the main approach for emissions reduction under the Kyoto Protocol, and defined many principles for carbon accounting, such as that it is the difference in carbon gains between a project activity and business-as-usual which can be credited. Under the CDM, methods under

Afforestation/Reforestation (A/R) project activities allow for blue carbon ecosystems and the VCS method VMO0033 refers back to the underlying A/R tool 14. This method considers trees and shrubs, but available default values are not directly applicable for saltmarsh vegetation present in South Australia.

The international market mechanisms are transitioning to the Paris Agreement, which will be implemented from 2020. The respective article 6.4 of the Paris Agreement, which seeks improved market mechanisms to replace the CDM (Herr et al. 2017), still requires international agreement on rules and procedures. These rules will cover issues of double-counting and international credit transfers, which will be of interest for Australian carbon market mechanisms as well.

Potential double-counting is currently preventing the use of voluntary carbon markets using international standards in Australia, as the Australian policy is not to cancel Kyoto units to avoid double-counting, as would be required by Verra, which manages the VCS. Double counting could arise if the same abatement is accounted for under both Australia's NGGI reporting and exported offshore by a third party scheme. If an activity is not included in the NGGI, double counting should not be an issue and credits from the voluntary market possible.

5.2 Assessment of alternative ERF methods for mangrove trees

As a method for blue carbon under the ERF was not available during the project timeframe, registration with an existing methodology was trialled. Mangroves >2 m in height and with 20% canopy cover are meeting the definition of forest under the forest category in Land-use, Land-Use Change and Forestry (LULUCF) of the IPCC Guidelines (IPCC 2006). The Human-Induced Regeneration of a Permanent Even-Aged Native Forest (HIR) was thus considered as a possible method for mangrove reforestation, and the tidal trial registered with the Australian Government's CER in August 2017.

The CER did not endorse the determination, as FullCAM (Full Carbon Accounting Model) is not able to model mangroves. In particular, the FullCAM guidelines are not allowing to add a model for mangrove as a tree species. A further issue was that the project for the trial pond reconnection had already acquired funding and begun to be implemented, thus not meeting the 'newness' requirement of the *Carbon Farming Initiative Act 2011* (see Appendix C.1).

In the absence of a blue carbon method under the ERF, the use of the HIR method has remained a possibility, although it only covers above-ground biomass, not the carbon stored in below-ground biomass and soil of mangrove. Using the HIR method for mangrove would thus intrinsically give less credit for carbon offset gains than are actually arising from a project activity. Similarly, the Native forest from managed regrowth (NFMR) method is meant for stopping activities inhibiting regeneration of native vegetation, such as land clearance. Both methods are thus not really suitable for mangrove for reasons of eligibility and modelling restrictions.

Here, we are providing a brief overview of the key aspects of the HIR method and NFMR method, as per the most recent guidelines (CER 2019). General aspects of the HIR method are compiled in Appendix C.2.

The HIR and NFMR are modelled methods, using FullCAM or Reforestation Management Tool (RMT) to determine the carbon stored. The approach aims for a Tier 3 level assessment, thus requires substantial data and evidence from field measurements to inform the model. The CER is cognisant of data limitations, which should not inhibit initial project stages, and expects that uncertainty will be reduced with further data collection.

Projects under the HIR or NFMR method are usually large in size to gain carbon offset from terrestrial forests, from ~500–300 000 ha, with a median project area of about 15 000–20 000 ha. This is a project scale which may be difficult to realise for mangrove. Furthermore, the CER expects that only areas that have forest potential are included as carbon estimation areas (CEAs), which would exclude areas of saltmarsh or mangrove <2 m in height, as well as creeks. This expectation does not fit the heterogenous and transient landscape of tidal wetlands.

Stratification under the HIR and NFMR refers to the demarcation of CEAs, and exclusion of areas with pre-existing forest cover or without forest potential, and a required step for the modelling. At the time of stratification, a CEA must have forest potential, evidenced by the presence of sufficient trees (seedlings or saplings) which can grow to >2 m in height. Initial stratification is expected to be based on field sampling, and checks for map accuracy. Further assessments can use remote sensing or other techniques (as approved by the CER) to detect change in forest regeneration. At the time of forest cover assessment, 90% of the CEA must have achieved forest cover to obtain a carbon offset certificate. All forest cover is to be mapped at the 0.2 ha scale.

The new guidelines for the HIR and NFMR include standardised approaches for mapping vegetation and monitoring regeneration of forest, and expect use of consistent methodology during the project. Field validation plots are to be used for training remote sensing classification (including field validation) and obtaining regeneration evidence. The types of data to be collected for survey plots are: photographs; location coordinates; woody species present that could exceed 2 m; stem density of each species, and the height of any species expected to meet 2 m at maturity to the nearest 0.5 m. It is further recommended to record standard forestry variables such as species biomass, canopy cover, age-class, and diameter at breast height. The recording of these data in the field is possible for mangrove, and respective data have been determined in the course of the project from mangroves in the reference areas (see chapters 3 and 4), and a chronosequence at Swan Alley (Clanahan 2019).

The database for mangrove forests in South Australia has grown through this project and further recent studies on the Swan Alley chronosequence. We now have dry mass and carbon estimates for above and below-ground biomass, litter and dead wood, know how tree density changes with forest age and the growth rate for *Avicennia marina*. Such data could be useful for developing a sub-model in FullCAM for *A. marina* mangrove. However, expectations of the HIR method are too focussed on terrestrial forests and the method does not appear suitable for blue carbon projects. Instead of a possible mangrove sub-model in the HIR method, the development of a blue carbon method under the ERF should be progressed.

5.3 Suitability of tidal reconnection project activity for the ERF

The introduction of tidal flow emerged as the highest ranking activity for blue carbon benefits in a national assessment of blue carbon enhancement activities for a possible method under the ERF (Kelleway et al. 2017a). This activity can include removing levee banks, installing tidal gates, or changing culverts, which increase tidal flow through changes in water regulation structures. The tidal gate installed at the trial pond for this project thus fitted fully into this activity.

We used a similar approach to Kelleway et al. (2017a) and the same series of questions for a suitability assessment of the tidal trial. Findings from the project, expert knowledge and associated projects (e.g. the chronosequence at Swan Alley) were considered in the replies. The initial assessment is given in Table 12, followed by an abatement integrity assessment (Table 13). The outcomes for the tidal reconnection of the salt pond gave a score ≥8, and further detailed assessment carried out (Table 14).

The assessment demonstrates the potential for tidal reconnection of salt ponds as a blue carbon project activity under Australia's ERF. The activity meets formal requirements, and yields carbon offset gains into the future, especially when applied at larger scale, together with additional ecosystem services, as explored in the following chapter.

Table 12: Initial assessment of suitability of tidal reconnection in salt ponds for the ERF, following the approach by Kellway et al. (2017a).

ASSESSMENT CRITERIA	OPPORTUNITIES FROM SALT POND RECONNECTION
1. Blue carbon enhancement activity scope	
1.1 Describe the specific blue carbon ecosystem activity that could enhance abatement.	Re-introduction of tidal flow into a salt pond from ceased industrial operation at the Dry Creek salt field. The installation of a tidal gate was successfully trialled and followed by passive revegetation with saltmarsh which commenced several months after reconnection. Abatement through introduction of tidal flow will be sequestration and avoided emissions, through an increase the biomass carbon pool, increased stock and sequestration into sediments, and reduced GHG fluxes to the atmosphere from the periodically flooded soils. Some of the increase in soil C originated from allochtonous seagrass detritus.
1.2 List the circumstances or conditions under which the activity is to be implemented.	Re-introduction of tidal flow could be applied to a series of further ponds in the salt field. The activity can also be considered for areas where tidal flow has been disconnected, e.g. where coastal wetlands were blocked off by levy banks or other coastal infrastructure. It can be considered in combination with planning for sea level rise and habitat retreat.
1.3 Where available, provide background information about the abatement activity. This could include case studies that demonstrate the successful implementation of the abatement activity.	The tidal gate installation at Dry Creek has proven to be a suitable engineering solution to reconnect previous salt ponds without harmful effects to the environment. In Swan Alley (Barker Inlet), reconnection occurred in the late 1930's when levy banks were abandoned. The successful revegetation show an increase in biosequestration as documented through a chronosequence.
2.Opportunity for uptake and genuine abatement	
2.1 Identify potential participant groups for the blue carbon enhancement activity.	Participant groups include, for example, Government (all levels), land owners (private, corporate), traditional owners, etc. Land ownership has to be clarified for possible activity areas, and early consultation held with stakeholders.
2.2 Estimate the potential volume of abatement for the blue carbon enhancement activity, taking into account scale of abatement over land mass area.	The areas for tidal reconnection of salt ponds could extend to about 2000 ha at the Dry Creek salt field.
2.3 Consider the extent to which the enhancement activity could have adverse social, environmental or economic impacts.	The activity of reconnecting salt ponds could have adverse economic effects on industries if they had alternative commercial plans. Adverse effects could be changes in the density of the vegetation, with restricted flight paths and foraging areas for shorebirds on salt field ponds.
2.4 Determine alternative measures (existing schemes, legislation etc.) that the enhancement activity could be (or already is) promoted through.	Tidal reconnection could be promoted through environmental management obligations or other conservation conventions and legislations. The installation of a tidal gate at Dry Creek was also a measure to remediate ASS which have built up in the salt ponds.
3 Additionality	
3.1 Demonstrate how emission reductions achieved through the blue carbon enhancement activity are unlikely to occur in the ordinary course of events.	For salt ponds, the ordinary course of events would be being permanently flooded with hypersaline water, and accruing ASS. On other coastal areas, reconnecting flow to land for revegetation is unlikely to happen without incentives to landowners. Alternative measures (point 2.4) could reduce the additionality.

Table 13: Abatement integrity assessment for re-introduction of tidal flow for mangroves and tidal marshes in South Australia. The integrity requirement formulation and scoring criteria are based on Kelleway et al. (2017a). Scores for each integrity requirement item are based on the tidal trial evaluation.

INTEGRITY REQUIREMENT	SCORING CRITERIA	SCORE	SCORE JUSTIFICATION
1. Undertaking the blue carbon enhancement activity must result in carbon abatement that is unlikely to occur in the ordinary course of events.	0. The enhancement activity is likely to occur regardless of ERF participation. 1. Based on available course of events information it is not possible to ascertain the likelihood of the activity occurring in the ordinary course of events. 2. -Based on available information, including current practice and existing regulations, it is considered likely that undertaking the activity would be additional to what is likely to occur in the ordinary course of events.	2	Tidal reconnection would not occur under ordinary course of events (salt field operation), the activity is additional.
2. Estimating the activity's carbon removals, reductions or emissions must be achieved using an approach that is measurable and capable of being verified.	0. There are currently no recognised measurable or verifiable approaches available to determine carbon removals, reductions or emissions relating to the activity. 1. There are measurement approaches but they are not currently backed by substantiated evidence. 2. There are recognised measurable or verifiable approaches backed by peer reviewed literature and validated case studies	2	Recognised measurable or verifiable approaches exist through international blue carbon method manuals (Howard et al. 2014) and VCS method VM0033. Peer-reviewed literature exists on effects of tidal reconnection on wetlands in the Hunter region (e.g. Howe et al. 2009).
3. Carbon abatement using in ascertaining the carbon dioxide net abatement amount for the activity must be eligible carbon abatement in accordance with the approach outlined in footnote 2.	0. Carbon abatement from the activity is not eligible carbon abatement. It cannot be counted towards Australia's NGGI. 1. It cannot be determined if carbon abatement from the activity is eligible carbon abatement. It is uncertain whether the carbon can be counted towards Australia's NGGI. 2. Carbon abatement from the activity is eligible carbon abatement and can be counted towards Australia's NGGI.	Mangrove: 2 (Biomass); 1 (Soil) Saltmarsh 1 (Soil)	Mangrove biomass carbon may be included under current forest carbon inventory (LULUCF) for trees >2 m height. For soils, Australia has commenced to include for stocks in the inventory, but there is still uncertainty about what can be counted towards the NGGI.
4. The approaches used for the activity must be	0. There is currently limited or nil clear and convincing evidence to support the	1 (can become 2 once publication are out)	Field investigations from this project have been carried out following standard methods.

INTEGRITY REQUIREMENT	SCORING CRITERIA	SCORE	SCORE JUSTIFICATION
supported by clear and convincing evidence	blue carbon enhancement activity. 1. There is supporting evidence but it is not considered to be clear and convincing evidence. 2. The proposed blue carbon enhancement activity and associated measurement approaches are supported by clear and convincing evidence backed by peer reviewed literature and validated case studies.		These studies can give support and evidence on the blue carbon enhancement activity. Manuscripts are in preparation. Further data are available from a chronosequence at Swan Alley over a decadal scale restoration after reconnection. Scientific literature on tidal reconnection activities has documented biosequestration benefits (e.g. Howe et al. 2009).
5. Material amounts of GHG that are emitted as a direct consequence of the activity must be considered.	0. Any material amounts of GHG emitted through the activity would be unable to be unaccounted for. 1. It cannot be determined whether there will be material amounts of GHG emitted as part of the activity 2. There are demonstrable approaches for ensuring material amounts of GHG will be able to be accounted for and deducted from net abatement amounts in carrying out the activity.	1–2	There are methods available to quantify emissions, including any material amounts that may occur. Measurements on GHG were done, and methane emissions emerged as negligible. Areas where 'leakage' could arise have not been studied for GHG emissions.
6. Estimates, projections or assumptions regarding activity abatement are conservative	0. Estimates, projections or assumptions used to work out the net abatement amount are not conservative. 1. It cannot be determined whether estimates, projections or assumptions are conservative but the approaches are anecdotally considered conservative. 2. - Estimates, projections or assumptions used to work out the net abatement are supported by peer reviewed literature that demonstrates conservativeness.	1–2	The estimates, projections or assumptions were based on measurement from project trial taken, and compared with values from references areas and scientific literature. Conservative estimates are given (e.g. considering soil C only to 30 cm), and mean values with ranges used. Manuscripts are in preparation.
	Total score	Mangrove (biomass) = 9 - 11 Mangrove (soil) = 8 - 10 Saltmarsh (soil) = 8 - 10	

*Footnote 2: *To be eligible carbon abatement, the abatement needs to be able to be captured* in *Australia's nationally reported GHG emissions. In the absence of current national reporting on blue carbon capture and storage, consideration should be given to the IPCC 2006 Guidelines (Volume 4 - AFOLU), and the 2013 Supplementary guidelines on wetlands (Chapter 4 Coastal Wetlands).*

Table 14: Detailed assessment of suitability of tidal reconnection in salt ponds for the ERF, following the approach by Kellway et al. (2017a).

ASSESSMENT CRITERIA	OPPORTUNITIES FROM SALT POND RECONNECTION
5. Identifying the baseline	
5.1 Specify a process for identifying the blue carbon enhancement activity baseline.	For salt pond reconnection activity, repeated measurements taken at adjacent ponds which are still operated under business-as-usual, allow a characterisation of the baseline.
5.2 List and justify the assumptions and uncertainties on which the baseline is based.	Baseline measurements have to be taken with sufficient effort, over several sites and times (see Kelleway et al. 2017a). For salt ponds, this includes sampling at several elevation strata, which can be difficult while a pond is still under business-as-usual operation. If literature values are used, they have to be for similar environmental settings, site conditions and uses, and more conservative calculations are to be used.
5.3 Describe the steps and/or processes involved in undertaking the abatement activity and identify all emissions sources and sinks directly or indirectly affected by the activity.	For reconnecting salt ponds, installation of a tidal gate is a now successfully proven process, with some short-term and localised emissions arising from engineering activities. No effect on adjacent mangrove and saltmarsh was recorded from tidal reconnection. An alternative process for reconnection can be removal of levy banks at locations of former creeks, which are still apparent on bathymetry maps. Passive revegetation by mangrove and saltmarsh occurs with dispersal of seed and propagules. Active revegation of mangrove by planting has to align with permitted activities under the ERF method used.
5.4 List all emissions sources and sinks affected by the activity in the table below. Indicate whether the source or sink is to be included or excluded from the baseline or GHG assessment boundary and provide justification for any exclusions. Expand the table to include additional sources and sinks, as necessary.	Table 18 from Kelleway et al. (2017a) is in general applicable. Our project showed that saltmarsh aboveground biomass should also be included in the carbon pools
6. Activity Area	
6.1 Specify how the blue carbon ecosystem enhancement activity area and boundaries would be determined.	Digital elevation maps are available for coastal areas of SA. Land-use classifications, tenure and conservation classifications are available at government departments and through NatureMapsSA. Boundaries are further subject to pond size, elevation, and project specific, after consultation with stakeholders.
7. Estimating abatement	
7.1 Provide a summary of approaches on how to calculate baseline emissions and removals. For any uncertainties around these approaches, outline what the uncertainties are, whether they are material and how could they be addressed.	The approaches used align with methods laid out by the IPCC 2013 Wetlands Supplement and the Blue Carbon Manual (Howard et al. 2014). Biomass values and allometric equations were obtained from recent studies. Soil carbon stocks were sampled in mangrove and saltmarsh. GHG emissions have been measured in the field (localised and short-term) and through an Eddy Covariance tower for mangroves (at St Kilda). Considerations for uncertainties around approaches used have been summarised by Kelleway et al. (2017a) and are also applicable in SA.
7.2 Provide a summary of approaches to calculate project activity emissions and removals. For any uncertainties around these approaches, outline what the uncertainties are, whether they are material and how could they be addressed.	Considerations given under 7.1 apply here as well. For project areas, finer scale resolution of surface elevation may be needed.

7.3 Provide a summary of approaches to calculate net GHG abatement. This should be the difference between the baseline and project activity emissions and removals.	For baseline and project scenarios, soil and biomass carbon pools were used, following internationally and nationally recommended methods. The abatement gain from the difference between project activity and baseline was calculated for several scenarios (see section 6.3).
7.4 Provide a summary of approaches on data collection methods for the baseline emissions and removals and project activity emissions and removals.	See 7.1
8 Double counting	
8.1 Provide a summary of approaches on how to avoid the double counting of up-stream and down-stream carbon sources that are already being captured in inventory reporting (e.g. carbon that enters the blue carbon ecosystem through river system or catchment area).	Double counting can arise from above ground biomass of seagrass being counted for the seagrass while alive, and then as carbon when deposited as wrack in mangrove and saltmarsh soils. This issue about allochtonous material is occurring throughout southern Australia and procedures to account for it are under consideration.
9. Permanence and Leakage	
9.1 Provide an assessment of factors likely to influence permanence (over both 25 and 100 year periods) of the carbon stored as a result of the blue carbon ecosystem enhancement project activity. Outline likely leakages that may eventuate through long term events, environmental or otherwise.	Permanence can be affected by sea level rise, in combination with land subsidence, and other coastal development infrastructure inhibiting landward retreat of coastal ecosystems. Along the coastline of the northern Adelaide plains, retreat opportunities exist and in combination with land use planning, any constraints to permanence through coastal squeeze can be managed in advance. Severe weather events or fire are unlikely to affect coastal wetlands in SA. A die-back of some saltmarsh species (*Tecticornia arbuscula*) around southern Australia of yet unknown cause could reduce the ABG and BGB of this taller shrubby saltmarsh. Leakage from salt pond reconnection is assumed to be low, as the salt field is not populated, and no dwellings or roads affected.
10. Monitoring and reporting	
10.1 Outline the elements of the activity that will be monitored and reported and describe how monitoring and reporting approaches will be undertaken, including frequency of monitoring and standards of monitoring.	Monitoring requirements are outlined in national and international methodologies for blue carbon (see Kelleway et al. 2017a, or VM0033). Measurements are similar to those for estimating abatement (see 7). Monitoring should follow consistent methods over time, with frequency decreasing over the duration of the project once abatement changes have stabilised. Over decadal timescales for monitoring projects, government agencies could assure the continuity and data base records.
11. Land ownership and legal right to carbon	
11.1 Outline land access and ownership rights issues that may affect the person who intends to carry out the activity through the ERF.	The Dry Creek salt field is a combination of privately-owned land and Crown land. Discussions with land owners have to occur and written agreements made about who owns the legal rights to the carbon credits. Ownership considerations have to consider longer time scales and that some land may become water in coming decades.

5.4 Summary of issues and outlook for blue carbon offset mechanisms

In the absence of a suitable method under the ERF, the trial pond reconnection was not eligible to be registered as an eligible carbon offsets project with the CER. The HIR method, which was used for the attempted registration, has emerged as an unsuitable method for blue carbon projects, as it would omit a large part of the carbon pool in soils and avoided GHG emissions, and has expectations which cannot be realised by the very nature of blue carbon ecosystems. However, if the FullCam model was updated to include mangrove forests, there may be opportunities to release the AGB. This could be a suitable pathway forward in appropriate landscapes. Lessons have been learned for any future renewed attempt at registration, including timing, as registration with the CER has to be made before a final investment decision.

Following approaches used in exploring potentials for a blue carbon method under the ERF, the assessment for the tidal trial has shown that tidal reconnection meets the offset integrity standard and further assessment criteria set by the Government. Carbon offset market mechanisms are currently in a period of transition, both nationally and internationally, which can bring uncertainty for project activities. However, it also offers the opportunity to inform developments at state, national, and international level. The introduction of tidal flow through tidal gates is a viable method for a project generating carbon offsets, and this project will be a relevant pilot for a national method.

6 Co-benefit analyses and up-scaling

Tidal reconnection of ponds in the Dry Creek salt field can not only have benefits for tidal wetland restoration and associated carbon offset opportunities, but can also enhance other ecosystem services, including social well-being, and bring possible economic returns. This chapter comprises an overview of co-benefits and ecosystem services associated with tidal wetlands in the region, the socio-economic context, and results from a survey of social and cultural values. The potential outcomes of tidal reconnection of further salt ponds are evaluated in a series of scenarios, upscaling the findings from the study to a possible larger scale.

6.1 Co-benefits and other ecosystem services

6.1.1 WHAT ARE CO-BENEFITS?

Globally, governments and policy-makers are working with science, business and the wider community to improve socio-economic wellbeing for all. This is clearly setup in the United Nations (UN) Agenda 2030, which is widely known as the UN Sustainable Development Goals (SDGs). While poverty alleviation, reducing hunger and access to education and health opportunities are key to achieve global goals for humanity, mitigating climate change by GHG reduction and responding to global warming through multilateral agreements such as the Paris Agreement can also facilitate achievement of the SDGs. Therefore, the benefits of addressing climate change are more than reducing GHG emissions and removal of excessive carbon di-oxide (CO_2) from the atmosphere. Such additional benefits of carbon reduction are termed co-benefits, or are referred to as ancillary benefits, side benefits, secondary benefits, collateral benefits, and associated benefits (IPCC 2001). These are described in the Assessment Report 3 of the Intergovernmental Panel on Climate Change (IPCC 2001) as:

> *The benefits of policies that are implemented for various reasons at the same time – including climate change mitigation – acknowledging that most policies designed to address greenhouse gas mitigation also have other, often at least equally important, rationales (e.g., related to objectives of development, sustainability, and equity).*

Co-benefits are also of importance for the feasibility of carbon offset options in South Australia.

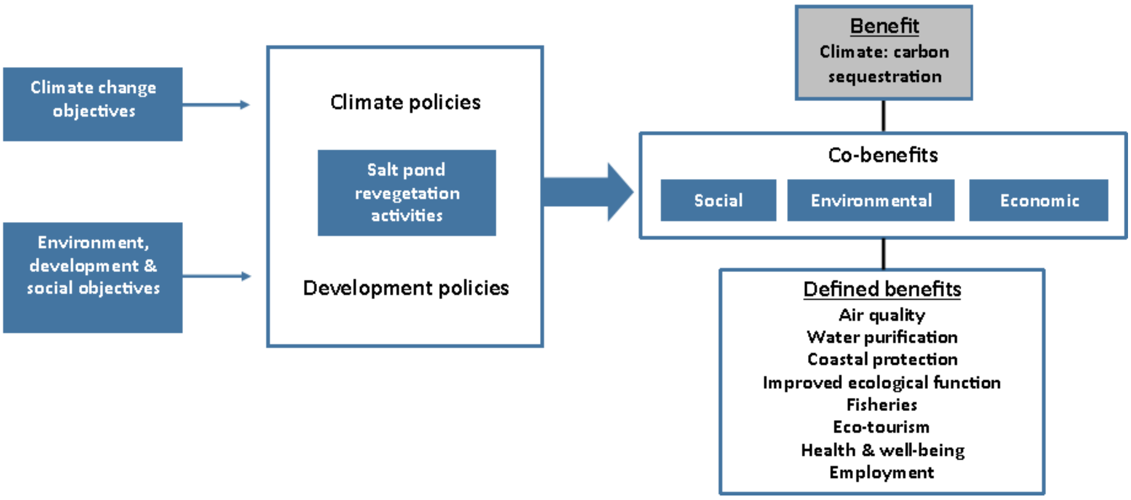

Figure 27. Conceptual overview how co-benefits can be derived through climate mitigation policies and activities, as for the exemplary case of salt field restoration. Examples of defined benefits within social, environmental or economic co-benefits are given. Concepts are adapted from the IPCC (2001).

Climate mitigation along with environmental, developmental and social objectives, are the basis for climate and development policies, which can support multiple co-benefits of restoration activities, as in the case of the Dry Creek salt fields (Figure 27). Here, we define co-benefits as positive additional outcomes of carbon sequestration by the establishment of mangroves and saltmarsh plants in the decommissioned salt ponds. In this case, carbon sequestration is the fundamental benefit with linked co-benefits broadly classified as (1) social, (2) environmental, and (3) economic (Figure 27). The co-benefits described here include more defined benefits such as: improved air and water quality; coastal protection; improved ecological functions such as fish nurseries; improved fishing opportunities; eco-tourism; improvement in mental health and well-being of residents; and employment in the nearby local government area.

6.1.2 CO-BENEFIT STUDY REGION

Co-benefits were assessed in a wider landscape context, for the area between Port Adelaide to the south, Thompson's Beach in the north, and Gawler in the east. The region falls under three local government areas (LGA); City of Playford, City of Port Adelaide Enfield and City of Salisbury (Figure 28). We included several major land cover classes in the three LGAs, as co-benefits of salt pond restoration activities are likely to flow-on to these areas. Further inland, the City of Playford area mainly consists of farming, agriculture, commercial and residential land cover. In comparison, the City of Salisbury and City of Port Adelaide Enfield have mainly commercial and residential land cover. Along the coastline, the land cover consists of mangroves and saltmarsh throughout, with salt evaporation ponds adjacent to the coastal wetlands (Figure 28). The marine habitats (e.g. seagrass beds) of the adjacent Gulf St Vincent to the west were also included. The assessment is structured around describing the socio-economic situation of the region and the current co-benefits and ecosystem services associated with the samphire coast. This assessment can inform on plausible co-benefits of salt field restoration activities.

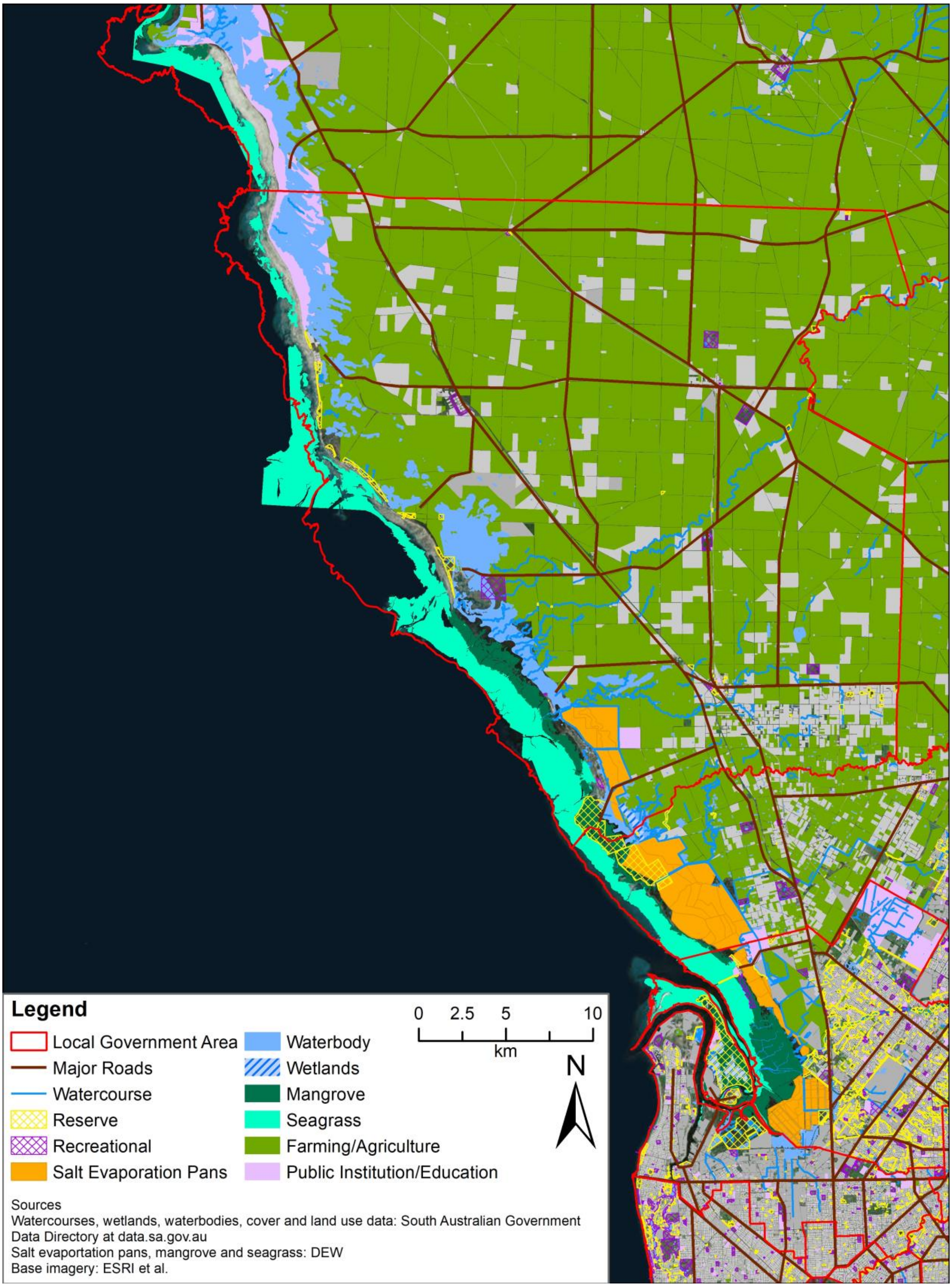

Figure 28. Area map of the co-benefit study region showing major land use classes. Approximate boundaries for three local government areas (City of Playford, City of Port Adelaide Enfield, and City of Salisbury) are displayed as red lines. Residential and commercial land cover are indicated by grey and white spaces.

6.1.3 SOCIO-ECONOMIC SITUATION

The three LGAs considered are important for their economic contribution to South Australia's Gross State Product, but vary with regards to socio-economic value indicators. For example, the Gross Regional Product (GRP) varies from $2709 million (City of Playford) to $8021 million for the City of Port Adelaide Enfield, which is closest to the city of Adelaide's port where manufacturing is a main source of employment (Table 15). Human population is larger where there is a dominant residential land use (i.e. also indicated by number of building approvals) as opposed to agricultural land uses further north along the coast in the Playford LGA (Table 15). The rate of population growth in the three LGAs lies above the annual population increase for the entire state. Across the three LGAs, human population growth is a sign of expanding residential suburbs, particularly in the Playford LGA (Table 15). The index of relative socio-economic disadvantage (SEIFA) was lower for the Playford LGA, which may be explained by employment opportunities in the region in health care and social assistance jobs (Table 15). Combined the LGAs have a population of >7000 Aboriginal and Torres Strait Islander people (Table 15). The coastal ecosystems in the three LGAs provide for fisheries and tourism. The Adelaide Dolphin Sanctuary and the Adelaide International Bird Sanctuary National Park - Winaityinaityi Pangkara attract large visitor numbers every year.

Table 15: Socio-economic value indicators for the three LGAs within the co-benefit region of the Dry Creek salt field. Data based on ABS 2018, NEIR 2016. M=million

SOCIO-ECONOMIC VALUES	PLAYFORD	SALISBURY	PORT ADELAIDE ENFIELD
Human population 2017	92,191	142,024	124,731
Human population growth (2011-2017)	11,507	9,144	8,239
Aboriginal and Torres Strait Islander population 2016	3,092	1,293	2,807
Gross regional product (GRP) 2016	$2,709 M	$6,151 M	$8,021 M
Main employer 2018	Health care and social assistance (22%)	Manufacturing (20 %)	Manufacturing (16 %)
Building approval value 2016	$170 M	>$500 M	$320 M
Tourism and hospitality value 2016	$110 M	$128 M	$141 M
Socio-economic disadvantage (SEIFA index) 2018	855	917	936
Main land use type	Agriculture	Residential	Residential

6.1.4 CO-BENEFITS AND ECOSYSTEM SERVICES OF THE REGION

The assessment builds on the premise that the benefit of carbon sequestration by restoration activities in the salt ponds also leads to social (e.g. cultural, recreational, tourism), economic (e.g., employment, food, fish, income) and ecological (e.g. biodiversity, clean air, freshwater, ecosystem services) co-benefits that are far-reaching. Healthy and well-functioning ecosystems contribute to human well-being through the provision of multiple ecosystem services (MEA 2005). In the study area, the key coastal ecosystems are seagrass, mangroves, and saltmarshes.

Globally, coastal ecosystems represent about 4% of total land area and over one-third of the world's population live along these coastlines (UNEP 2015). Marine and coastal ecosystems are among the most productive, providing about 63% of the total economic value of all ecosystems (Barbier et al. 2011; Costanza et al., 1997, 2014; Liquete et al. 2013).

In South Australia, the northern Adelaide coastline of the Gulf St Vincent has a diverse range of marine and coastal ecosystems with high conservation value, recognised by the recent declaration of the Adelaide International Bird Sanctuary National Park – Winaityinaityi Pangkara (Caton et al. 2009; Fotheringham and Coleman 2008; Lamanna et al. 2017). This is, however, affected by ongoing economic development which can impact on ecosystem services. The services provided by the coastal ecosystems north of Adelaide have recently been valued at $3.5 billion annually based on a benefit-transfer model using global values (Sandhu et al. 2018). These values may increase over time (Sandhu et al. 2018).

Ecosystem services can be broadly classified into categories of provisioning, regulating and maintenance, and cultural services (de Melo Soares 2018; Englund et al. 2017; Newton et al. 2018). This study used the four classifications to evaluate the ecosystem services provided by the coastal habitat in the study region (Table 16). Attributes considered follow either the Millennium Ecosystem Assessment (MEA 2005) or The Economics of Ecosystems and Biodiversity (TEEB 2010) classification of ecosystem services. The Common International Classification of Ecosystem Services (CICES, v4.3 or v5.1) (Haines-Young and Potschin 2018) was used to define classes within attributes for ecosystem services, in conjunction with further recent reviews and classifications of ecosystem services (Czúcz et al. 2018; Englund et al. 2017). The attributes, CICES classes and local values together explain the social, environmental and economic co-benefits (Table 16; Figure 27).

Based on previous ecosystem service estimates of the Barker Inlet and Port River region (Sandhu et al. 2018), local values were assessed with reference to supporting literature (Table 16). No valuation of ecosystem services is as yet available for Australia's tidal wetlands (Himes-Cornell et al. 2018), apart from several values in the global data base (Appendix D, Table D.1). Valuation for ecosystem services can include measures such as replacement costs, or willingness to pay assessments, derived from surveys as has been done for coastal ecosystems elsewhere (Needham and Hanley 2019; Newton et al. 2018). Further studies are needed for a valuation of local ecosystem services. As mangroves are encroaching on saltmarsh with sea level rise (Saintilan et al. 2019) and the local landscape context is changing, links between ecosystem structure and function and ensuing services need to be assessed (Barbier 2019, Kelleway et al. 2017b). However, information about the benefits of ecosystem services exists in global assessments and recent reviews (Czúcz et al. 2018; de Melo Soares 2018; Newton et al. 2018; Smith et al. 2017). Information on particular local ecosystem services is available from scientific literature cited in this project (see chapters 3, 4), in particular for social aspects (e.g. human use and value of the region from surveys, section 6.2). Where no published information was available to support the local ecosystem services, findings from unpublished projects, student theses, general expert knowledge of the area and online sources were considered. (Table 16).

Table 16: Ecosystem services and co-benefits provided by seagrass, mangroves and saltmarsh in the three local government areas (LGA) adjacent to the Dry Creek salt field. CICES = Common International Classification of Ecosystem Services, v.5.1 and 4.3. The 'local value' includes examples for benefits from respective services in the study region.

ECOSYSTEM SERVICES CATEGORY	ATTRIBUTE	CICES CLASS	LOCAL VALUE	REFERENCE
Provisioning (biotic)	Food	Wild plants (terrestrial and aquatic, including fungi, algae) used for nutrition	Mangrove honey; bushfood	Clarke (2013) https://www.sanativefoods.org.au/samphire; https://www.willungafarmersmarket.com.au/our-producers/artisan-condiments/do-bee-honey
	Raw materials, medicinal resources	Fibres and other materials from wild plants for direct use or processing (excluding genetic materials)	Bio-chemicals, natural medicines	Clarke (2013); Lorbeer et al. (2013)
		Wild animals (terrestrial and aquatic) used for nutritional purposes	Recreationally & commercially caught seafood	Bloomfield and Gillanders (2005); Connolly (1994); Jackson and Jones (1999); McArthur and Boland (2006)
	Genetic materials	Seeds, spores and other plant materials collected for maintaining or establishing a population	Seed harvest for plant propagation	
	Habitat provisioning	Wildlife shelter, nursery sites	Biodiversity of native mammals, birds, reptiles, fish & invertebrates	Caton et al. (2009); Coleman and Cook (2009a,b) Coleman et al. (2017); Cribb et al. (2013); Dittmann et al. (2012); Edgar and Shaw (1995); Edyvane (1999); Kemper et al. (2008); Lamanna et al. (2017); McArthur and Boland (2006); Payne and Gillanders (2009); Vogt (2019)
	Biological productivity	Foraging habitats, nursery for fish and marine invertebrates	Migratory shorebirds; maintaining nursery populations	Coleman and Cook (2009a); Close (2008); Lamanna et al. (2017); Purnell et al. (2015)
Regulation & Maintenance (Biotic)	Water purification, air quality regulation	Bio-remediation by micro-organisms, algae, plants, and animals	Water quality regulation, aesthetics; nutrient cycling, decomposition	Fotheringham and Coleman (2008); Fox et al. (2007)
		Filtration/sequestration/storage /accumulation by micro-organisms, algae, plants, and animals	Carbon sequestration	Ch. 3, this report
		Smell reduction	Decomposition of algal mats	Duong (2008)
		Noise attenuation	Low noise environment	
	Erosion regulation	Control of erosion rates	Stabilization of soils and sediments	Cann et al. (2009)

ECOSYSTEM SERVICES CATEGORY	ATTRIBUTE	CICES CLASS	LOCAL VALUE	REFERENCE
	Regulation of water flows, regulation of extreme events	Hydrological cycle and water flow regulation (Including flood control, and coastal protection)	Flood prevention, wave attenuation, coastal protection from storms, flooding and sea level rise	Fotheringham and Coleman (2008); Fox et al. (2007)
	Pollination	Pollination (or 'gamete' dispersal in a marine context)	Habitat for native pollinators, plant, diversity of native pollinators	Clarke and Myerscough (1998); Hermansen et al. (2015)
	Biological control	Seed dispersal	Regeneration of vegetation	Ch. 4, this report
	Biological control	Maintaining nursery populations and habitats (Including gene pool protection)	Nursery habitats for recreationally & commercially useful fisheries species	Bloomfield and Gillanders (2005); Connolly (1994); Jackson and Jones (1999)
		Pest control (including invasive species)	Biological interactions reducing pest and invasive species	
	Water regulation	Regulation of the chemical condition of salt waters by living processes	Health of coastal and marine ecosystems	Nayar (2015)
	Climate regulation	Regulation of chemical composition of atmosphere and oceans	Sequestration of carbon in coastal wetlands (blue carbon), mitigation of global warming	Ch. 3, this report
Regulation & Maintenance (Abiotic)		Mediation by other chemical or physical means (e.g. via Filtration, sequestration, storage or accumulation	Biogeochemical mediation of waste, toxics and other nuisances in estuaries	
		Mass flows	Sediment stabilization, Natural barrier from storms, sand bars providing coastal protection from flooding and sea level rise	Cann et al. (2009); Fotheringham and Coleman (2008); Fox et al. (2007)
		Liquid flows	Erosion control, Flood protection from natural levees	
		Maintenance and regulation by inorganic natural chemical and physical processes	Regulating living conditions, e.g. sea breezes	
Cultural	Recreation & ecotourism	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	Sport and fitness, nature-based recreation, relaxation	Ch. 6.2 this report
		Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	Eco-tourism, recreation, fitness; de-stressing or mental health	Ch. 6.2 this report
	Knowledge systems and	Characteristics of living systems that enable scientific	Research, knowledge about the environment and nature	Caton et al. (2009), Edyvane (1995)

ECOSYSTEM SERVICES CATEGORY	ATTRIBUTE	CICES CLASS	LOCAL VALUE	REFERENCE
	educational values, cultural diversity, aesthetic values	investigation or the creation of traditional ecological knowledge		
		Characteristics of living systems that enable education and training	Skills or knowledge about environmental management, citizen science programs, community engagement, coastal ambassadors	https://www.naturalresources.sa.gov.au/adelaidemtloftyranges/get-involved/volunteering/volunteer-opportunities/become-a-coastal-ambassador; https://www.naturalresources.sa.gov.au/adelaidemtloftyranges/coast-and-marine/coast-and-marine-ecosystems
		Characteristics of living systems that are resonant in terms of culture or heritage	Local identify, cultural history	Caton et al. (2009); Ch. 6.2 this report
		Characteristics of living systems that enable aesthetic experiences	Inspiration for culture and arts	Oakley (2005); Caton et al. (2009); https://estuary.org.au/the-estuary/
	Spiritual and religious values	Elements of living systems that have symbolic, sacred or religious meaning	Spiritual importance, mental well-being, social cohesion	Ch. 6.2 this report; EBS (2016)
		Elements of living systems used for entertainment or representation	Documentaries, nature films	https://www.youtube.com/watch?v=1WokMcl2yIw; https://www.youtube.com/watch?v=NG09j52oe6Q
		Characteristics or features of living systems that have an existence, option or bequest value	Conservation parks, Marine Parks, Aquatic Reserves, Adelaide Dolphin Sanctuary, Adelaide International Bird Sanctuary – Winaityinaityi National Park; mental and moral well-being	Ch. 6.2 this report

An important ecosystem service for the Gulf St Vincent region is recreational and commercial fishing. Four species that are commonly targeted by recreational and commercial fishing sectors are; Blue Swimmer Crab (*Portunus armatus*), Western King Prawn (*Melicertus latisulcatus*), King George Whiting (*Sillaginodes punctatus*) and Southern Garfish (*Hyporhamphus melanochir*). Collectively, the commercial catch of those species contributes >$39 million to South Australia's economy annually (Table 17). Mangroves and seagrass ecosystems within the Barker Inlet region adjacent to the Dry Creek salt field are known as important habitat for these four species, particularly as nursery for their juvenile stages (Bloomfield and Gillanders 2005; Connolly 1994; Jackson and Jones 1999). Recreational catches of King George Whiting and Southern Garfish exceed commercial catches. The recreational fishing effort in Gulf St Vincent accounts for 28% of all of the state's total recreational fishing effort (Giri and Hall 2015).

Table 17: Recreational and commercial fishing indicators for the Gulf St Vincent region. Species listed are the most targeted fished species that depend on seagrass and mangroves in the Port River and Barker Inlet region as nursery, adjacent to the Dry Creek salt field. Recreational catch values for 2013–14 obtained from Giri and Hall (2015). Commercial catch in tonnes for 2017 obtained from FRDC website https://fish.gov.au (2019) and commercial values obtained from Status of South Australian Fisheries report 2012–13 (PIRSA 2015). GSV = Gulf St Vincent; NGSV = northern Gulf St Vincent; SA = all of South Australia. M = million.

FISHING	CATCH IN TONNES/ YR		
TARGETED SPECIES	RECREATIONAL	COMMERCIAL	COMMERCIAL VALUE/YR
Blue Swimmer Crab	65 NGSV	218 GSV	$2.5 M GSV
Western King Prawn	-	237 GSV	$30 M SA
King George Whiting	50 GSV	46 GSV	$4.8 M SA
Southern Garfish	94 NGSV	68 GSV	$1.8 M SA

While ecosystem service valuation does not represent a direct economic value, Payments for Ecosystem Services (PES) are becoming an increasingly more common measure (Locatelli et al. 2014; Salzman et al. 2018). Under PES, a 'supplier' of ecosystem services is linked with a 'user' who is a beneficiary of the ecosystem service, and paying for use in a market (Everard 2018). Applying PES to the blue carbon project requires further refinement of this instrument, but could possibly enable the bundling or stacking of carbon offset with co-benefits (Hejnowicz et al. 2015; Runting et al. 2016; Lau 2013).

6.2 Social and cultural values – assessing the cultural ecosystem services of the coastal wetlands between Torrens Island and Thompson Beach

It is increasingly acknowledged that finding solutions to environmental challenges requires consideration of the cultural beliefs or values of a social group in a given area (Jackson 2006; Pizzirani et al. 2014; Small et al. 2017; Verschuuren 2006). Environmental worldviews that shape individual and collective attitudes, beliefs and behaviours, at a range of scales, are culturally derived. Cultural values are rarely quantified or acknowledged however, on the basis of their intangibility (Millington et al. 2014). Unsurprisingly then, of the four ecosystem services, cultural ecosystems services (CES) are the least understood and least used despite their importance and relevance. This study makes a contribution towards understanding the social and cultural values associated with the coastal landscapes of the wetlands north of the city of Adelaide.

6.2.1 METHODS

There is no standard methodology to assess CES. Following the literature, this study used an inquiry-based approach to consider cultural values associated with human-environment interactions (Church et al. 2014; Fish et al. 2016; Greenaway et al. 2015). Three components frame the inquiry: practices (actions people take or things people do), and spaces (settings in which actions happen—places, landscapes or ecosystems), and ecosystem benefits accruing from the intersection of space and place (meanings or significance generated though specific practices in specific spaces) that contribute to well-being (mental and physical health benefits). Three elements give rise to cultural ecosystem benefits: identity, experience and capability (see Table 18).

Table 18: Elements that contribute to cultural ecosystem benefits. (Adapted from Church et al. 2014: p.19)

ELEMENT	DESCRIPTION
Identities	**Symbolic associations – sense of belonging** e.g. ecosystems play a role in the process of place identification through which ideas of affiliation and attachment develop
Experiences	**Encounters with nature (mental or physical); feelings** e.g. feelings of calm arising from encountering some physical attribute of ecosystems, or an experience of nature deemed aesthetically pleasing
Capabilities	**Acquisition of skills, proficiencies and health; wisdom, judgment, insight needed to prosper** e.g. Use of ecological phenomena in processes of knowledge acquisition for intellectual and scientific advancement; acquisition of personal skills and knowledge through which people may advance their situation in life (for example through acquiring gainful employment)

The views of 89 participants were collected in July 2018 through the administration of an online survey to clubs and interest groups and a household questionnaire delivered to 500 residences adjacent to the coastal region. The survey asked a mix of closed and open-ended questions. Whilst the data is not able to be generalised, people participating in the study articulated a cultural connection to the coastline north of Adelaide, providing decision-makers with an indication of the value of the region to people who both live near it or use it for specific purposes.

6.2.2 RESULTS

The study attracted a wide geographical spread of respondents. Respondents did not necessarily live in the region to have an attachment to it. More men (44) than women (35) participated in the study. Forty four per cent of respondents were over the age of 60, followed by those of middle age (between 41 and 60 years) (38%).

Visiting patterns—85% of respondents who answered the question, said they had visited the coastal region at least once in the last year and more than 40% visited at least weekly (see Appendix D, Figure D.1 and Figure D.2). The majority of respondents had a long engagement with the region; 47 (53%) of respondents said they visited the region for the first time more than ten years ago, and a further 13 (15%) at least six years ago.

Practices—this study asked participants what activities they undertook when visiting the coastal region. The most commonly selected activity was 'observing nature/scenic appreciation', followed by birdwatching (Figure 29). Respondents also identified a range of different action-oriented or sporting activities (walking, boating etc.) and pursuit of interests such as photography. The region also serves as a meeting place and a site for spiritual and traditional activities for some. Almost one-third of respondents (27 of 89) said they belonged to a community or social group that used the coastal region between Torrens Island and Thompson

Beach. Respondents were given seven statements by which to rate various attributes of the region. The statement that 'the coastal region encourages healthy living' achieved the second highest rating with 86% of respondents rating this as either 'important' or 'very important'.

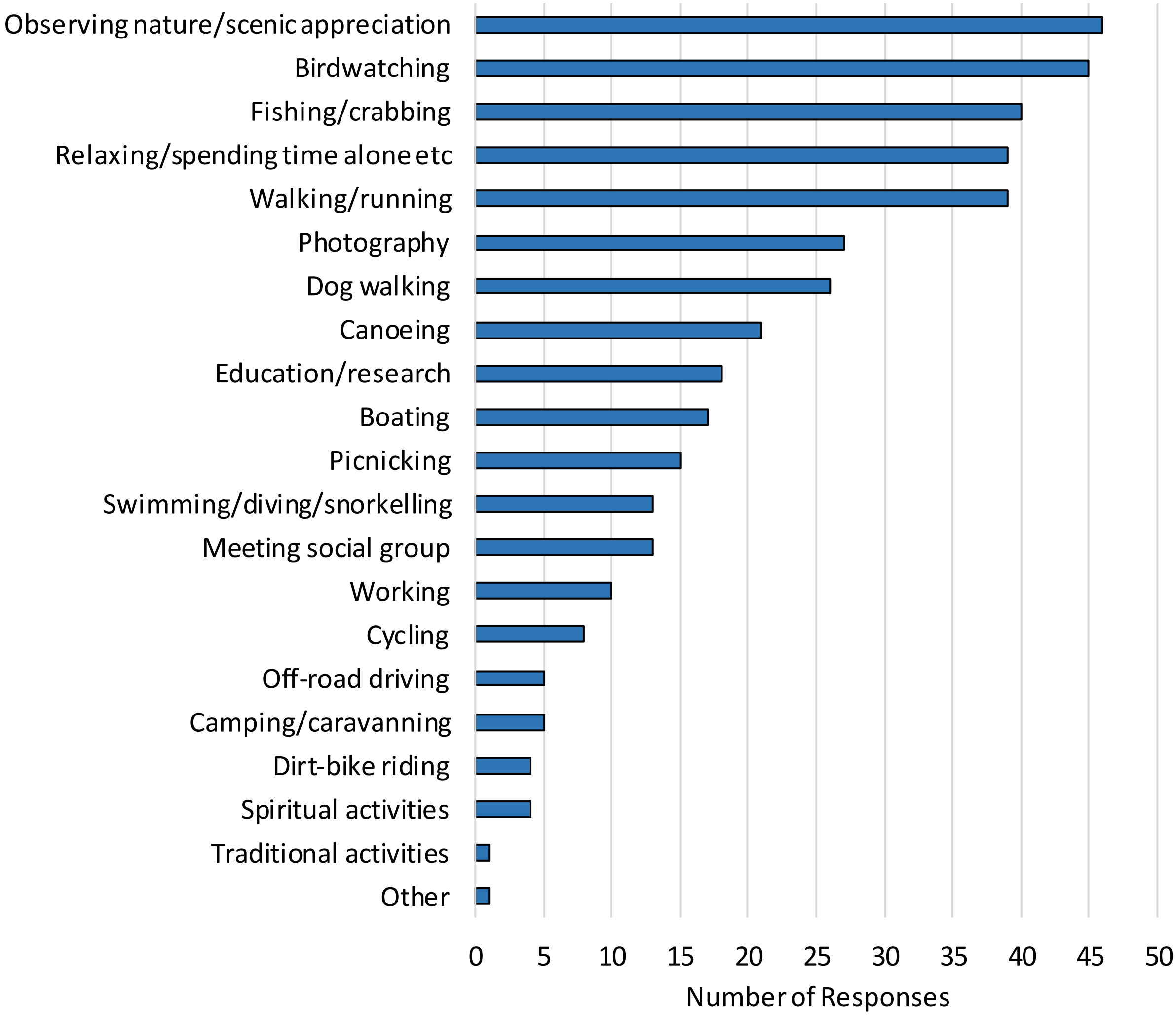

Figure 29. Practices-Activities undertaken in the study region (by number of responses).

Spaces—the surveys asked people about 11 of the most prominent sites within the region representing a mix of spaces from those under formal protection to commercial and non-commercial sites (see Appendix D, Figure D.2 and D.3). The awareness of and visits to particular sites corresponds to the interests and activities carried out by respondents. Overall, protected spaces and non-commercial sites were the places respondents identified as having visited most. The three commercial sites, of all the places identified in the survey, represented places that had been heard of but never visited by the largest number of respondents. Respondents were given nine statements by which to rate what the coastal region signified to them personally (see Appendix D, Figure D.4). The statement that 'the coastal region has places I enjoy visiting' achieved the highest rating with 92% of respondents rating this as either 'important' or 'very important'.

There was a correlation between visitation and importance rating. Respondents are more likely to rate sites as being important if they have visited the site. For example, the Adelaide International Bird Sanctuary National Park - *Winaityinaityi Pangkara*, was one of the most frequently visited of all 11 sites and was also rated as one of the most important sites. When asked about the attributes of the coastal region as 'providing places for family connection' and 'providing places for people to meet', 80% of respondents who answered these questions rated 'space' elements as being 'important or very important' (see Appendix D, Figure D.5).

Ecosystem benefit—identity element – When asked if they had a personal cultural connection to the region, several respondents expressed their 'identity' attachment to place by talking about their family and/or ancestral connections and about the place as being 'home'. When respondents were asked directly about their sense of belonging, or, if they had an attachment to the coastal region, several respondents identified their length of the connection to the place (rootedness), and place as a site of memory, and reminiscence. Of the nine personal attribute statements (see Appendix D, Figure D.4), 'the coastal region helps generate good memories for me', achieved the second highest rating with 88% of respondents rating this as either 'important' or 'very important'. More than three quarters of respondents rated the statement 'the coastal region fosters a sense of pride in me' as being 'very important' or 'important'. An example response related to the identify element was:

> *I have lived at Parham for almost 40 years. I love the whole coastline; the changing seasons, birdlife, fishing and crabbing. I visited the beaches with my parents who are now no longer here. I have many memories to share with the next generation for them to enjoy [ID43 Online Survey]*

Ecosystem benefit—Experience element – When asked what benefits the coastal region offered them, many respondents in this study gave 'experience' examples describing feelings such as calmness and/or peacefulness arising from encountering a physical attribute of this coast, and, happiness generated from an aesthetically pleasing experience. More than 80% of respondents rating the nine personal attribute statements (see Appendix D, Figure D.4), rated the 'the coastal region has places where I go to relax' as being either 'very important' or 'important', which was also reflected in statements like:

> *I feel at home under the open bowl of the sky, listening to the birds, squelching in the mud and not hearing another human or human activity in the entire sound-scape. It's what makes the metropolitan area bearable for me [ID40 Online Survey]*

Ecosystem benefit—Capability element – Ninety-five per cent of those responding to the statement that the coastal region 'provides for knowledge (education/history/nature)' (see Appendix D, Figure D.5), rated this element as either 'very important' or 'important'. Respondents gave various examples of the capabilities they have developed on the basis of a cultural practice they carried out in the study region. Participants articulated the role that ecological phenomena played in shaping their individual and social capacities to help them understand, and encourage them to take action through monitoring, educating or conserving, stating for example:

> *There is such diverse beauty in these areas any time of year and the scope of birds, wildlife and flora is captivating. Increased involvement over the past three years with ongoing surveys and volunteer activities has endeared this wonderful and unique environment to me. [ID44 Online Survey]*

Respondents were asked to rate the importance of a number of assets in the region (see Appendix D, Figure D.6). 'Environmental Qualities (landscape, aesthetics and flora and fauna)' of the region were rated by almost 100% of participants as either 'very important' or 'important' and retaining this importance in the future. Similarly, 'recreational facilities- areas for playing/exercising' and 'educational opportunities- research, training & field observation' were rated as being 'very important' or 'important' both now, but even more so in the future.

Respondents were asked their opinions about possible future scenarios and concerns about possible impacts of a changing climate (see Appendix D, Figure D.7). Two-thirds of respondents either ‘agreed’ or ‘strongly agreed’ they were concerned about the impacts of erosion, flooding and/or storms affecting their use of the coastal region between Torrens Island and Thompson Beach. They used statements like:

> *If current features are threatened, they need to be protected. We have a responsibility to preserve as much as practicable. I am an infrequent visitor, however, I am familiar with more frequent users and their reliance on the area. It is inexcusable to do nothing, only to realise too late what should have been done. [ID21 Household Survey]*

The survey sought respondent’s preferences for the future of the region (see Appendix D, Figure D.8). Almost half (48%, n = 38) said they wanted the region to be preserved and to be more protected than it is currently. At the other extreme, only one person completing the question wanted to see extensive development in the region. One-third of respondents wanted sustainable development in the region, with the remaining 16 per cent (n = 14) wanting the region to stay the same.

This cultural values study illustrates the importance of the coastal wetlands north of Adelaide attributed to it by people living adjacent to it and by those who come from outside the region to make use of its spaces and places. The findings suggest that the coastal region between Torrens Island and Thompson Beach, as perceived and reported by respondents, is contributing to human well-being in significant ways. Its naturalness is highly valued. Many participants in this study would like to limit development in the region and to enhance environmental protection. The cultural values associated with the environmental spaces and cultural practices should therefore, be considered in future planning for the region along with the values of provisioning and regulating ecosystem services. There is considerable scope to further investigate the things that matter to the people in this region. The study has accessed groups of people who use the region for particular types of activities. The perspectives of those who may participate in gathering and consuming pursuits, and heavy impact activities such as off-road driving, are missing. The region is likely to be of value to such users and their input would provide a different perspective to that generated in the study presented here.

Key findings suggest that people value places they visit. Providing access to the region is therefore important. It has been observed that some key educational sites are currently underutilised (the Middle Beach Samphire Trail) or in disrepair (St Kilda Mangrove Trail). Given the importance rating of the region as a site of education, it is worth considering mechanisms to improve these facilities, both in terms of access and ongoing maintenance.

6.3 Spatial and temporal up-scaling of carbon offset opportunities from tidal reconnection of salt ponds

Restoring or protecting blue carbon ecosystems can be part of nature-based solutions to mitigate climate change (Cohen-Shacham et al. 2016; Vanderklift et al. 2019). Tidal reconnection constitutes a nature-based solution, generating carbon offset opportunities and supporting ecosystem services and societal benefits (Figure 30). Positive results from the ecological restoration of the salt ponds may motivate the tidal reconnection of more ponds. Here, we explore possible benefits from a larger scale tidal reconnection by extrapolating findings from the trial pond to a wider area. Ecosystem services cannot be easily scaled up without further modelling for transferring value estimates to a larger area (Brander et al. 2012). We therefore concentrate on the calculation of potential gains for carbon offset opportunities based on soil and biomass carbon data.

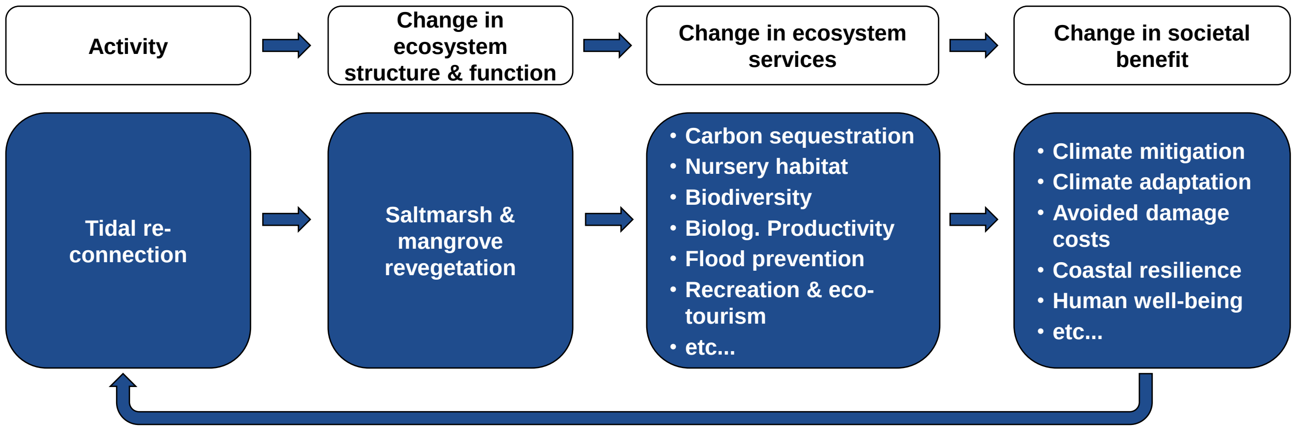

Figure 30. Conceptual overview on ecosystem service assessment and benefits obtained from tidal reconnection of salt ponds, following a nature-based solutions approach (graphic based on Arkema et al. 2017).

The reconnection of a salt pond was trialled for one of the smaller ponds of the Dry Creek salt field (see chapter 2), and data collated for the 1.5 years post gate opening. This short timeframe captures only early stages of recolonisation, but samples from reference sites gave data on late successional stages for saltmarsh and mangrove. Trajectories for recolonisation pathways were obtained from the seed dynamics experiments (see section 4.2), and further investigations on a chronosequence in nearby Swan Alley area (Clanahan 2019; Mosley, pers. comm., see also proof of concept report (Dittmann et al. 2019) for details). As data were collected from defined elevation strata and digital elevation maps (DEM) were available for the area (see section 2.3), it was possible to scale up in space. The resulting projections provide indications for the possible gains for blue carbon along the samphire coast.

6.3.1 SCENARIOS

To assess the carbon sequestration potential and co-benefits at different spatial and temporal scales, a baseline (business as usual) and three scenarios were considered; selected based on their hypothetical feasibility. Carbon offset calculations for each of the scenarios under various sea level rise predictions (following RCP2.6, RCP4.5 and RCP8.5 by the IPCC) are given in the proof of concept report (Dittmann et al. 2019). The various hypothetical scenarios are outlined below.

Business as usual: this scenario assumes the salt ponds are kept as salt ponds, and informs the baseline, i.e. no tidal reconnection occurring.

Scenario 1 – *Trial pond reconnected*: under this scenario, the trial pond XB8A stays reconnected and the development is modelled into the future.

Scenario 2 – *Low-lying ponds reconnected*: under this scenario, adjacent salt ponds, especially those with a single levee bank on the seaward side that could be opened up for tidal reconnection, were included in the assessment. These low-lying ponds mostly overlap with area that is crown land.

Scenario 3 – *Higher-lying ponds reconnected*: this scenario comprised the ponds from scenario 1 and 2 and added additional salt ponds located in the upper intertidal, and also ponds of the salt field area further north near Middle Beach.

The spatial extents for each scenario are outlined in Figure 31, showing that scenario 2 included mostly 'Mangrove-low marsh' and 'Tidal saltmarsh' strata, whereas higher elevations of the 'Supra-tidal saltmarsh' were added in scenario 3 (Figure 32a). The similarity in bathymetry and proximity to the gulf between the trial pond (scenario 1) and the ponds considered in scenario 2 (Figure 31), supports the underlying assumption that revegetation processes after tidal reconnection, and increases in carbon stocks for the soil and biomass carbon pools, will be comparable to findings from the tidal trial.

Based on digital elevation maps, the areas for each of the three strata, for which carbon data were assessed (see chapter 3), were calculated and the total area of each stratum determined (Figure 32a). For more accurate upscaling in each scenario and under future sea level rise (see Dittmann et al. 2019), the project area for each scenario was divided into six elevation strata. These strata corresponded to the three main vegetation classes (see section 2.3), with one sub-stratum (pure mangrove forest), and two additional classes for the areas that are at elevation above and below the vegetation class elevations.

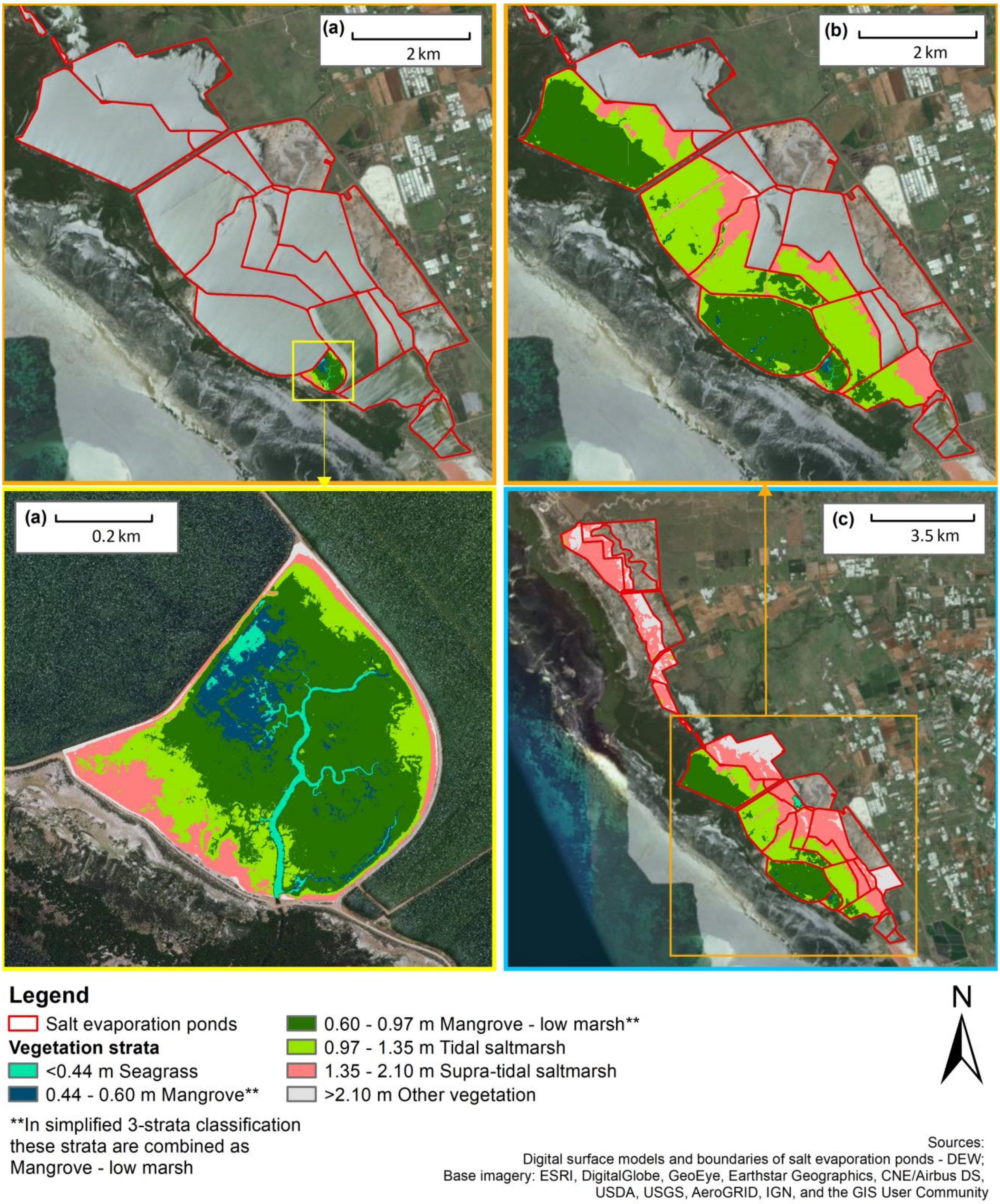

Figure 31. Maps of the samphire coast north of Adelaide outlining the boundaries and extent of vegetation strata considered under the scenarios: (a) scenario 1 – the trial pond stays reconnected (shown at two different scales); (b) scenario 2 – additional low-lying ponds are reconnected to tidal flow; and (c) scenario 3 – higher-lying ponds are reconnected in addition to those opened up under scenario 1 and 2.

Stratification was done based on DEMs of the pond area and surrounds (see section 2.3). It is important to note that outside the trial pond area, DEMs of the salt ponds were developed from point bathymetric surveys, and therefore lack detail of historic channels and possibly of low or high lying local areas in the ponds. The estimates of the areas of the vegetation strata that will re-establish at the lower elevations (mangrove and low saltmarsh) are therefore likely to be conservative.

Strata specific carbon stock values for soil and biomass (chapter 3, Table 10) were multiplied by the strata specific area under each scenario to provide initial estimates of potential total carbon gains. These estimates were further refined, and forecasting was made for carbon gains within 30 years.

In the absence of data on rates of growth and spread of saltmarsh species in the different strata, rapid spread and growth of saltmarsh species was assumed based on observed rapid colonisation of the trial pond. It was also assumed that there is little increase in carbon accumulation associated with saltmarsh biomass once these plants are established and fully grown. Carbon accumulation in mangroves was modelled based on the growth rates and area expansion rates of mangrove forest known from a nearby chronosequence (Clanahan 2019). The extent of mangrove forest was predicted on an annual basis, segmented by tree age, and associated biomass was determined. This was adjusted to suit the strata specific carbon stock values for this site (chapter 3, Table 10).

Potential increases in soil carbon stocks were calculated for each scenario based on assumed C-sequestration rates for each stratum, and the area of the strata. For the 'Mangrove-low marsh' stratum the mean organic C-sequestration rate calculated for the reference area using the constant rate of supply sedimentation model (chapter 3, Table 5) was used. As only a few of the sediment profiles were interpretable for sediment accumulation, no error estimates can be given for C-sequestration rates. Wilkinson et al. (2018) report minimum organic carbon accumulation rates for mangroves and coastal wetlands from a global analysis. These rates were used to obtain a generic ratio for C-sequestration in coastal wetlands relative to that in mangroves, and this ratio was applied to our C-sequestration rate in the 'Mangrove-low marsh' to estimate C-sequestration rates in the 'Tidal saltmarsh' and 'Supra-tidal saltmarsh'. For the business-as-usual baseline, a soil C stock measurement from the trial pond was used prior to reconnection and assumed to remain steady over time, based on no significant change in soil C stock measured over a 1.5 year period from a nearby control pond (Mosley et al. 2018).

6.3.2 THIRTY YEAR FORECASTING

The estimated forecasts of carbon gains within 30 years predict full establishment of all saltmarsh species for all scenarios, but full establishment of mangrove only for scenario 1. For scenarios 2 and 3, it is predicted that full establishment of mangrove will only occur at about year 66. There are therefore predicted to be further carbon gains for scenario 2 and 3 over and above those outlined for the 30-year forecast below. These longer term carbon gains are considered further in the proof of concept report (Dittmann et al. 2019) under permanence in light of sea level rise.

The estimated forecasts of carbon gains within 30 years of reconnection, indicate that once mangrove and saltmarsh are fully established in the respective areas under each scenario, the total carbon stock could potentially reach about 463 000 t CO_2e for the area of scenario 2, and over 652 000 t CO_2e for the area of scenario 3 (Figure 32b). Rates of carbon gains associated with biomass are initially expected to be rapid (Figure 33a) as saltmarsh plants spread and become established. Further gains are associated with growth of mature mangroves.

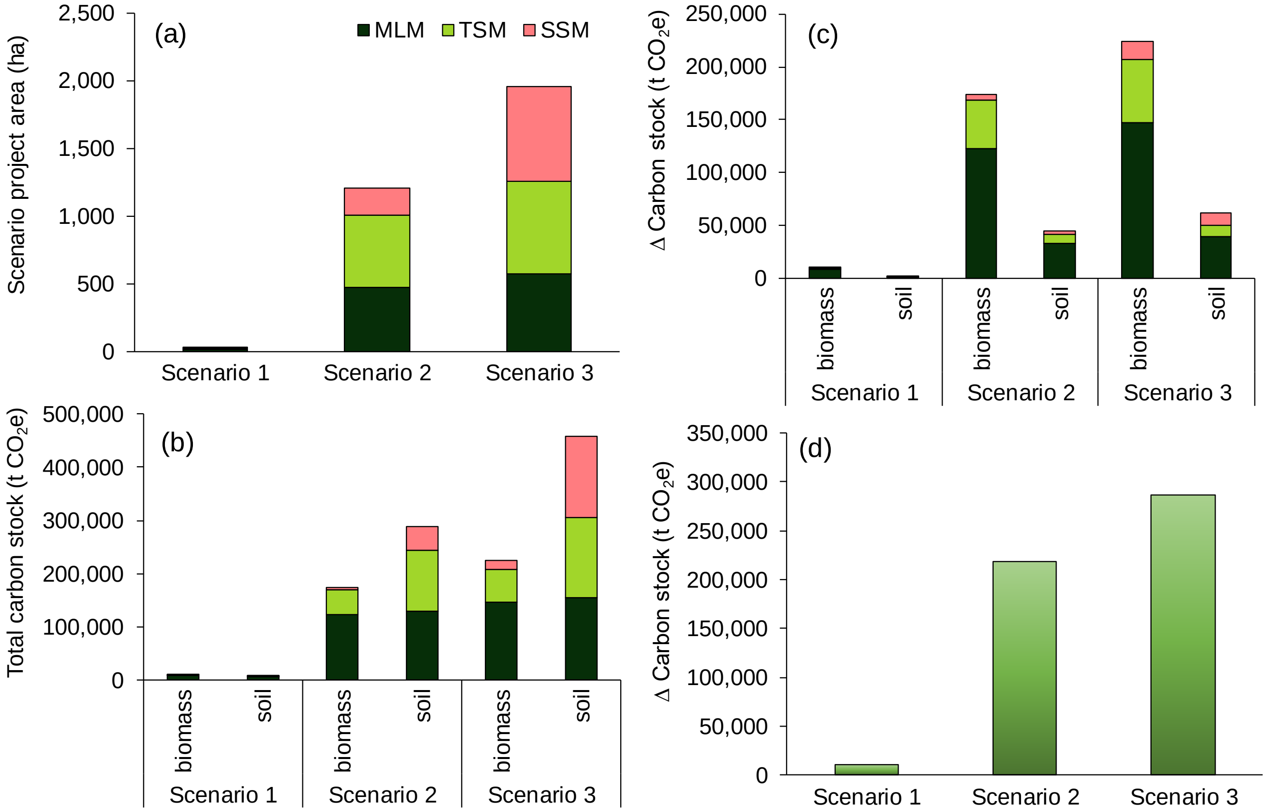

Figure 32. Changes in area and carbon stocks for different hypothetical scenarios. (a) Project area for each scenario considered, and breakdown of the area for each stratum; MLM = 'Mangrove-low marsh', TSM = 'Tidal saltmarsh', SSM = 'Supra-tidal saltmarsh'. (b) Estimated total carbon stock as t CO_2e for the respective area of each scenario, based on a 30 year timeframe without sea level rise. (c) Estimated net project benefit from the change in carbon stock as t CO_2e for the respective area of each scenario, based on a 30-year timeframe without sea level rise. The contribution of soil and biomass carbon pool is indicated separately, as scenario 3 includes mainly additional supra-tidal elevations with lower and sparse vegetation. (d) Total net project benefit for the combined soil and biomass carbon stock change subject to each scenario activity.

However, the net project benefit from the increase in carbon stock under the 'with- project' scenarios above the 'baseline' (business-as-usual for salt ponds), would be lower, as the baseline carbon stock (using the trial pond XB8A) was relatively high (Figure 33). The net increase in carbon above the baseline could be over 218 000 t CO_2e for the area of scenario 2, and over 250 000 t CO_2e for the area of scenario 3 (Appendix D, Table D.2). While soil and biomass carbon pools both increase, the largest net gains, approximately two thirds of total, result from biomass gain in previously unvegetated ponds (Figure 32c).

Taking the carbon price from the auction of the CER in December 2018 ($13.87), the potential gain in carbon stocks under the three scenarios in a project timeframe of at least 30 years could be over $150 000 for scenario 1, over $3 million under scenario 2, and over $3.5 million under scenario 3. These values are indicative only and subject to the assumptions made for this up-scaling. They can also be higher as the trading price for carbon increases over time. They are further explored with actual sea level rise and sea level rise predictions by the IPCC in the proof of concept report (Dittmann et al. 2019).

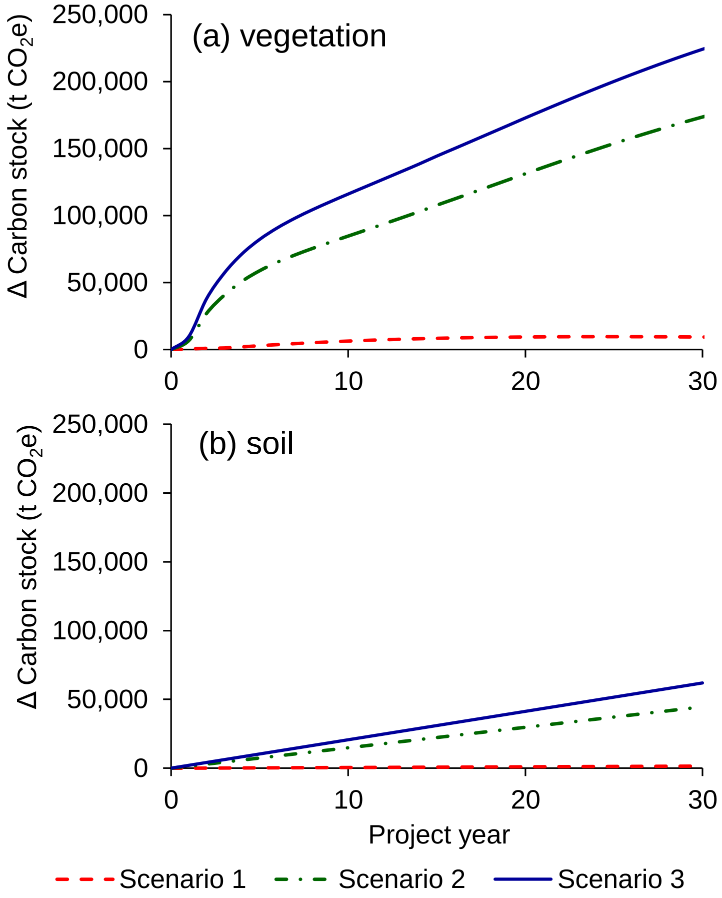

Figure 33. Estimated rate of accumulation of carbon stock as t CO_2e in the (a) biomass carbon pool and (b) soil carbon pool, for each scenario over a 30 year timeframe, without sea level rise. For comparability across the three scenarios with different areas, the plot shows the net increase in soil carbon gains compared to a baseline, set as zero as no change under business as usual.

6.4 Summary and discussion of blue carbon and co-benefits from tidal reconnection

Carbon sequestration from tidal reconnection is one of the benefits that can be derived from restoration of salt ponds, with benefits increasing the larger the restored area. As the project has shown, reconnection to tidal flow is a feasible activity, which generates elevated soil carbon concentrations as well as carbon stored in the saltmarsh vegetation establishing inside the pond. In addition to the blue carbon value, the habitat provided by restored saltmarsh and mangrove will increase provisioning ecosystem services, as well as ecosystem services which are regulating and maintaining benefits to humans. Furthermore, empirical data from the survey identified strong cultural ecosystem values for local residents and people from further afield in South Australia. Human well-being can be enhanced when salt fields are restored to natural tidal wetland ecosystems.

Environmental management prioritising single ecosystem services can result in lower co-benefits than approaches targeting multiple services (Atkinson et al. 2016). For the Dry Creek salt field, a broader scope for restoration, rather than focussing only on carbon offset opportunities, could generate and sustain multiple co-benefits. Coleman (2013) identified that remediating salt ponds to tidal wetlands would bring multiple benefits including increased biodiversity, carbon sink, room for habitat to retreat from sea level rise, storm surge buffer and coastal flood protection.

Such benefits were also considered in the planning stages for the Adelaide International Birds Sanctuary (DEWNR 2013). Restoring the salt field to tidal wetlands for multiple benefits would align with international examples, where mangroves and saltmarshes as living shorelines are seen as a ‘triple win’ for climate mitigation (carbon sequestration), adaptation (as a nature-based defence against sea level rise) and conservation (Davis et al. 2015; Narayan et al. 2016; Sheehan et al. 2019; Sutton-Grier and Moore 2016).

7 Translation of outcomes

The tidal reconnection trial has successfully demonstrated that opening a salt pond to tidal flow can lead to restoration of coastal wetland. Data show that even after just 1.5 years, the reconnected pond was naturally revegetated by saltmarsh, started to increase soil organic carbon content, and did not increase GHG emissions. As in other nearby tidal wetlands, seagrass detritus had been washed into the pond and accounted for a part of the increase soil organic carbon, as an allochthonous carbon component. Modelling projections for several hypothetical scenarios show the possible net benefit from the change in soil and biomass carbon stocks, which could be 218 146 t CO_2e in three decades for a scenario reconnecting about 1210 ha of low-lying salt ponds. Should more salt ponds be reconnected to increase the wetland area to about 1963 ha, the net project benefit could increase to 286 007 t CO_2e in three decades. In addition, ecosystem services, including social and cultural values, will benefit from restored coastal wetlands.

The project findings are of relevance as a proof of concept that tidal reconnection is a suitable project activity for restoration of a salt field, which can inform further decision making on future options of the Dry Creek salt field. The outcomes of the project will be presented to the relevant stakeholders to facilitate the decision process.

Results from the project have been presented to the scientific community at national conferences, and at workshops with DEW. Outcomes from the project were continuously made available for DEW through collaboration within the project team. Formal updates were presented at project advisory committee meetings and a briefing for the DEW Climate Change Group in September 2018.

Findings from the project and the demonstrated feasibility of the tidal reconnection activity have informed the current development of a blue carbon strategy for South Australia. In addition, the knowledge gained and shared during this process has substantiated the strategy development, and increased South Australia's reputation as an emerging leader in the blue carbon sphere.

Knowledge communication extended to the Commonwealth Department of Environment and Energy (DoEE), with a presentation given at a workshop on scoping a blue carbon method under the Emissions Reduction Fund (ERF) in March 2018. The importance of the project was further show-cased to the DoEE at a site visit in December 2018 and informed the progress in advancing the roadmap development for an ERF blue carbon method. Re-introduction of tidal flow is the top-ranking project activity to be assessed for a blue carbon method under Australia's ERF. The knowledge and experience gained from this tidal trial project will thus be of strategic importance for further developments in harnessing blue carbon.

The proof of concept for tidal reconnection as a blue carbon activity is further summarised in an accompanying report (Dittmann et al. 2019). A story map combining the outcomes will communicate the project to the wider public.

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Appendix A – Carbon pool data

A.1 SOIL CARBON DATA

Table A.1: Increase in soil carbon stock (t C ha^{-1}) for the project area over the study period, adjusted for the area of the three differentiated strata.

	AREA	CHANGE IN SOIL CARBON STOCK		
	ha	SEPTEMBER 2017	APRIL 2018	OCTOBER 2018
Mangrove-low marsh	21.27	1210	1459	1692
Tidal saltmarsh	5.95	275	535	450
Supra-tidal saltmarsh	4.87	396	360	180
Total	32	1881	2354	2322

A.2 SEDIMENT AND CARBON SEQUESTRATION DATA

Sediment accumulation models

The simplest model assumes constant sedimentation and constant depositional flux of $^{210}Pb_{ex}$ (Constant Flux Constant Sedimentation-CFCS). Since ^{210}Pb decays rapidly (22.3 years half-life) and most is sourced from atmospheric sources during sedimentation, the concentration of $^{210}Pb_{ex}$ decreases with depth. Equation 3 (Krishnaswamy et al. 1971) calculates sedimentation rate from excess ^{210}Pb decay:

$$\mathrm{Sed} = \frac{\mathrm{D_E}\lambda_{210}}{\ln^{\mathrm{A_B}}/_{\mathrm{A_E}}} \qquad (3)$$

Where Sed is the sedimentation rate (cm yr^{-1}), D_E is depth of A_E (cm), λ_{210} is the decay constant for ^{210}Pb, A_B and A_E are the measured initial and end excess ^{210}Pb activity (in Bq/kg or dpm/g).

The other age model used is referred to as the Constant Rate of Supply (CRS) model. The dating is based on the comparison of $^{210}Pb_{ex}$ inventories below a given depth with the overall $^{210}Pb_{ex}$ inventory in the sediment core. The accurate determination of the $^{210}Pb_{ex}$ inventories is of critical importance and required for the application of the CRS model (Appleby 2001). The age of sediments at certain depth can be calculated using the formula:

$$t = \frac{1}{\lambda}\, ln\left(\frac{A(0)}{A}\right) \qquad (4)$$

where *t* is age of the layer or time elapsed since formation , λ is the ^{210}Pb disintegration constant, A(0) is the total $^{210}Pb_{ex}$ inventory for the whole core, and A is the cumulative $^{210}Pb_{ex}$ inventory below a given depth. The CRS model is generally preferred and is used for carbon sequestration calculations. Nevertheless both CFCS and CRS age models were used to calculate sedimentation rates (Appendix A Figure A.1).

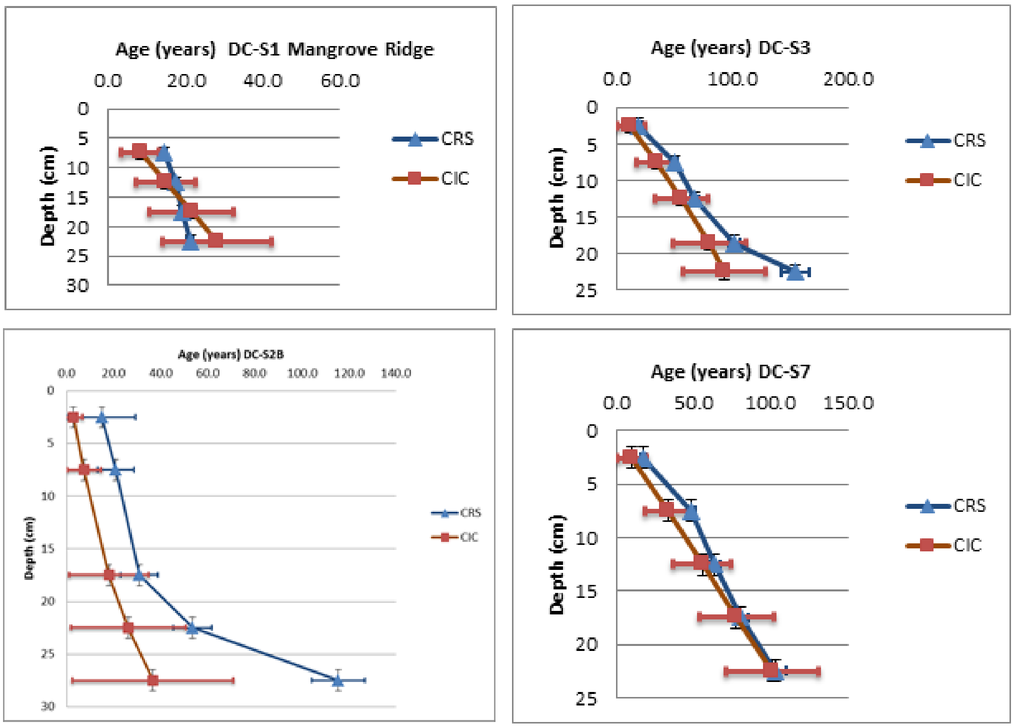

Figure A.1. Age-depth calculations for the four representative sites using both the preferred age model CRS (Constant Rate of Supply) and CFCS (Constant Flux Constant Sedimentation).

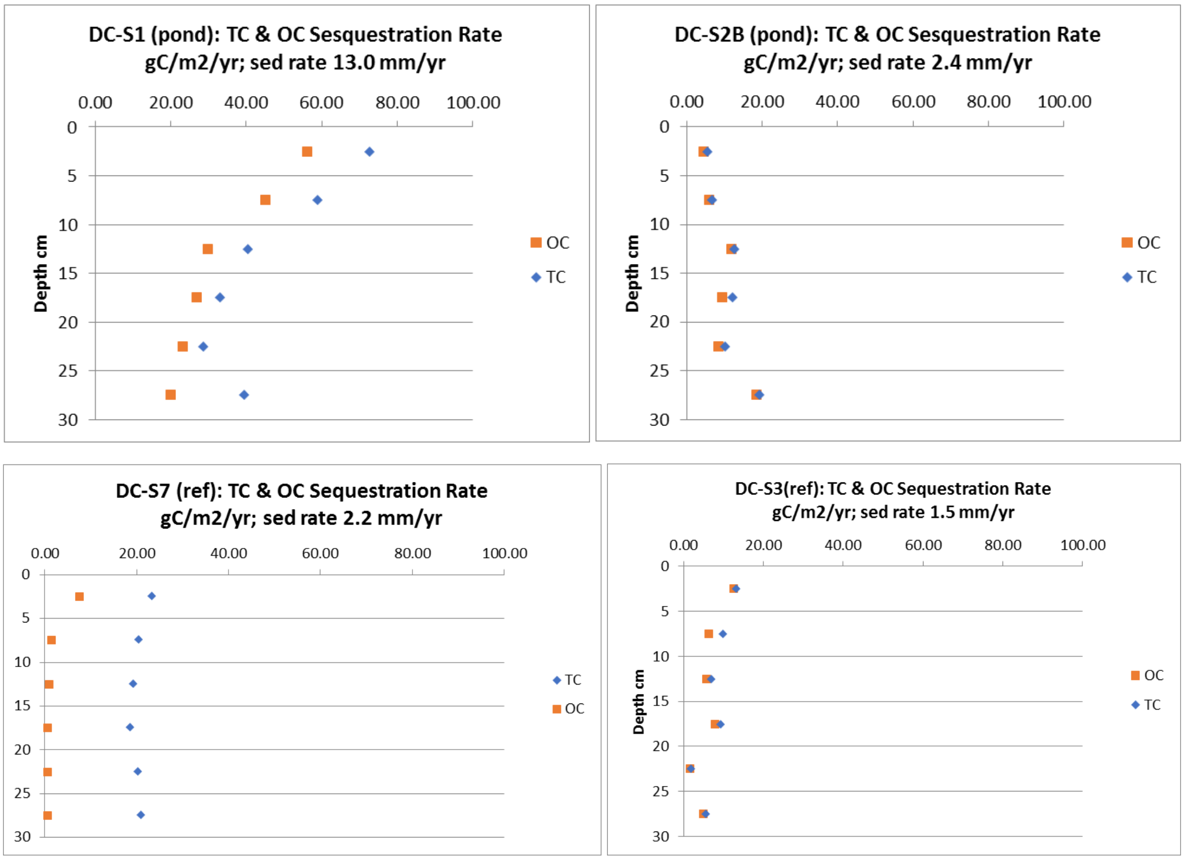

Figure A.2. Carbon sequestration rates for total carbon and organic carbon for the drained pond sites DC-S1 and DC-S2B and reference sites DC-S3 and DC-S7.

A.3 GREENHOUSE GAS MEASUREMENTS

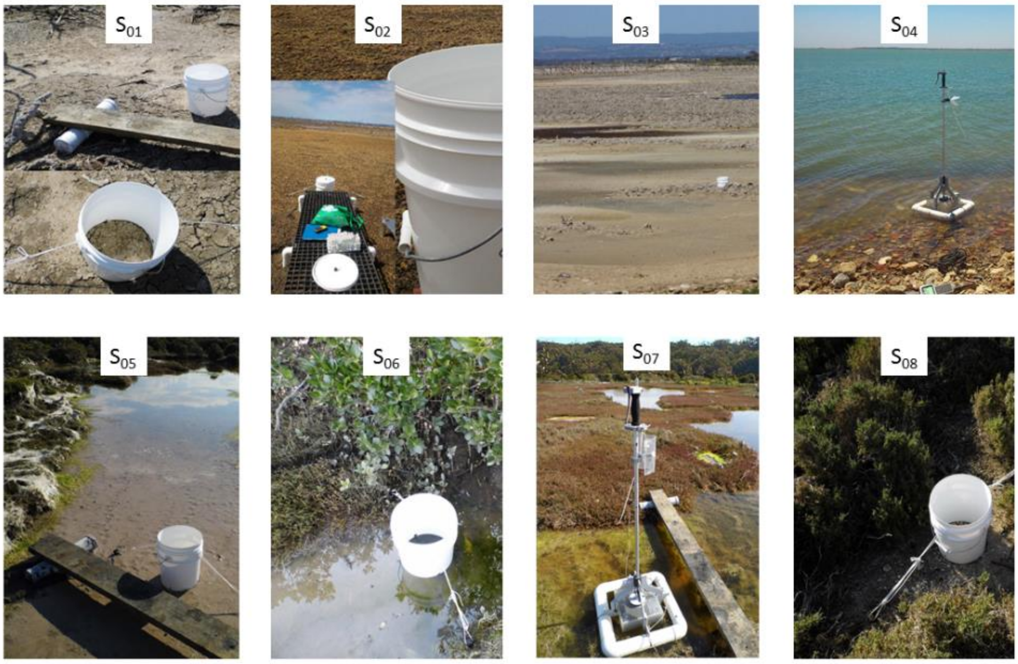

Figure A.3. Greenhouse gas sampling locations (S01-03 in Pond XB8-A, S04 in the control pond, S05-08 in the reference site), showing the sampling locations distributed in three different strata, and tidal channels. (i) < 0.6 – 0.97 m 'Mangrove-low marsh', S01 and S06; (ii) 0.97 – 1.35 m 'Tidal saltmarsh', S02 and S05; (iii) 1.35 – 2.1 m 'Supra-tidal saltmarsh', S03 and S08; and (iv) tidal channel S07 (and S09 not shown in the trial pond).

Table A.2: Summary of greenhouse gas flux measurements undertaken over the project period at sampling sites in the trial pond, control pond, and reference area. The different symbols indicate different methods used (X bucket chambers, * aquatic chambers, + LI-8100).

	TRIAL POND XB8A				CONTROL POND	REFERENCE AREA			
SAMPLING DATE	S01	S02	S03	S09	S04	S05	S06	S07	S08
26/09/2017	X	X	X		✶	X	X	✶	X
8/11/2017	X	X	X	✶	✶	X	X	✶	X
15/05/2018	+	+	+	+	+	+		+	+

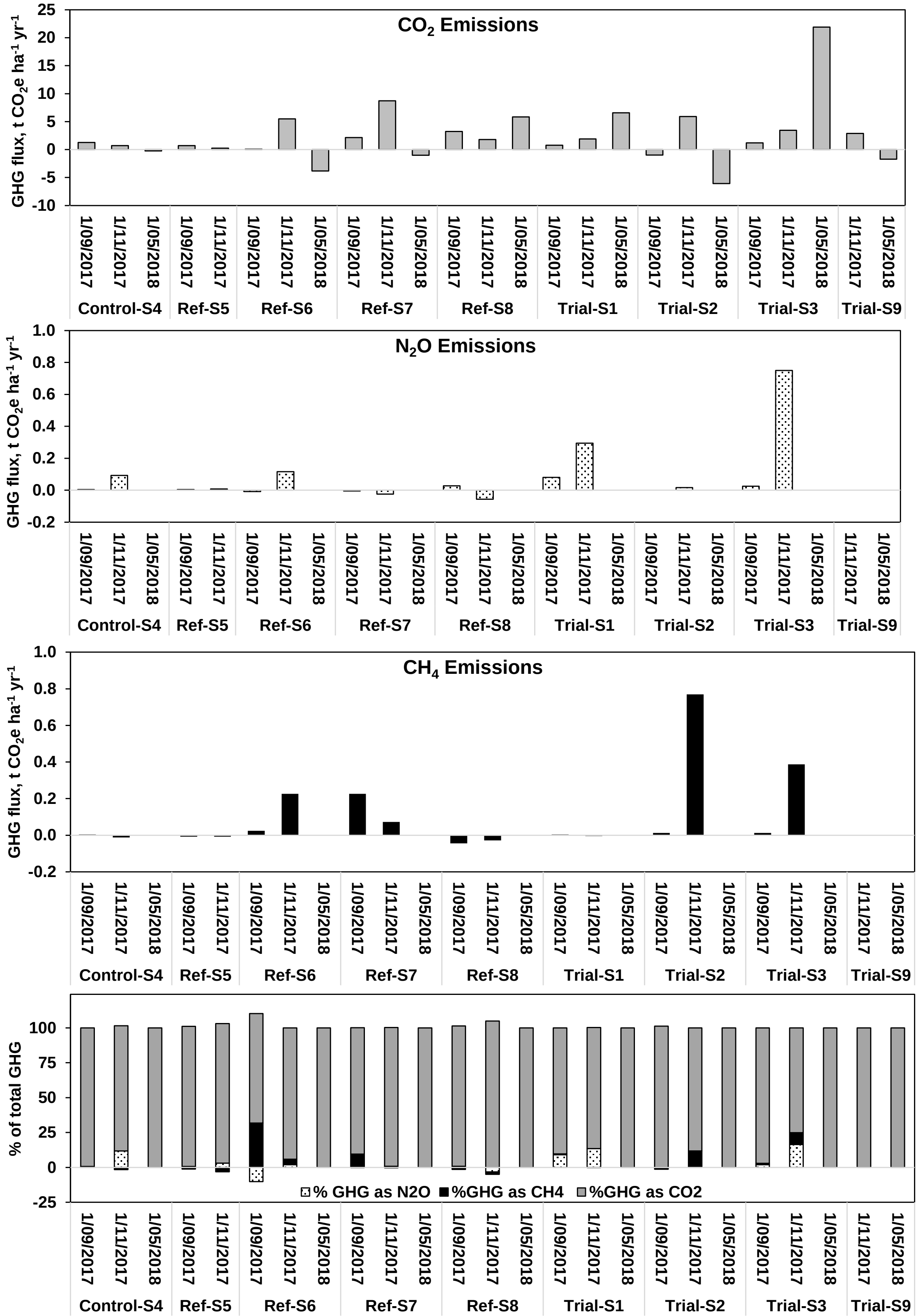

Figure A.4. Fluxes of CO_2e (average ± SE) for CO_2, N_2O and CH_4 and % of total greenhouse gas flux across the sampling areas at three occasions. Note that the method of measurement changed between the first two and the last measurement in May.

A.4 BIOMASS CARBON DATA

Carbon stocks in mangrove and saltmarsh vegetation (section 3.5) were determined in the area adjacent to the trial pond and the reference area, and at each of these two sites for the three strata/vegetation types (section 1.2) (Figure A.5). Biomass and carbon stocks for the various vegetation carbon pools were significantly different across the strata, but not between sites, apart from higher root biomass and carbon at the adjacent area in the 'Tidal saltmarsh' (Table A.3). Mangrove pneumatophores accounted for only a small part of the carbon stock, and this respective pool was significantly higher at the reference site.

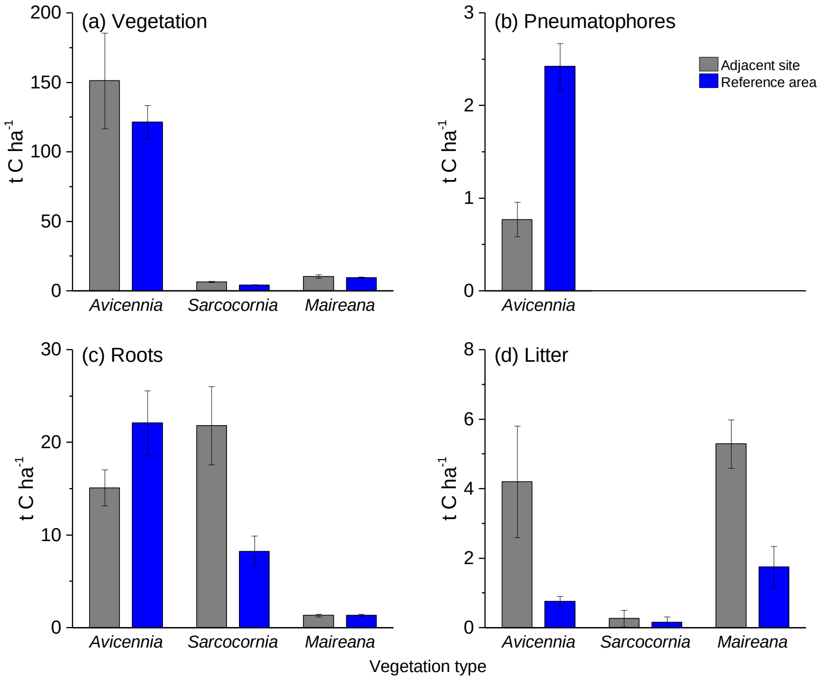

Figure A.5. Carbon pools (mean ± 1SE) of (a) vegetation, (b) pneumatophores, (c) roots and (d) litter for the vegetation types: *Avicennia marina* mangrove forest, *Sarcocornia quinqueflora* saltmarsh, and *Maireana oppositifolia* shrubland, separated by the area adjacent (outside) of the trial pond, and a reference area near St Kilda.

Table A.3: Test outcomes for two-way ANOVA for biomass and carbon stock data for the main carbon pools across the two sites and three strata. Pneumatophores were only found in the mangrove stratum. P-values indicating significant differences are highlighted in bold.

			BIOMASS			CARBON STOCK		
Carbon pool	Source	df	SS	F	*P*	SS	F	*P*
Vegetation	Site (Si)	1	0.09	10.74	0.1049	0.06	7.33	0.1675
	Strata (St)	2	22.10	149.27	**0.0001**	16.19	160.66	**0.0001**
	Si x St	2	0.02	0.12	0.9213	0.02	0.16	0.8856
	Res	18	1.33			0.91		
Roots	Site (Si)	1	0.08	0.22	0.6693	0.05	0.22	0.6638
	Strata (St)	2	7.29	98.44	**0.0001**	4.52	107.85	**0.0001**
	Si x St	2	0.73	9.83	**0.0012**	0.45	10.76	**0.0006**
	Res	18	0.67			0.38		
Litter	Site (Si)	1	1.02	6.87	0.1972	0.58	7.29	0.1707
	Strata (St)	2	5.36	23.51	**0.0002**	3.61	26.11	**0.0002**
	Si x St	2	0.30	1.30	0.3065	0.16	1.15	0.3474
	Res	18	2.05			1.25		
Pneumatophores	Site (Si)	1	0.30	22.92	**0.0297**	0.21	23.83	**0.0296**
	Res	6	0.08			0.05		

Appendix B – Revegetation

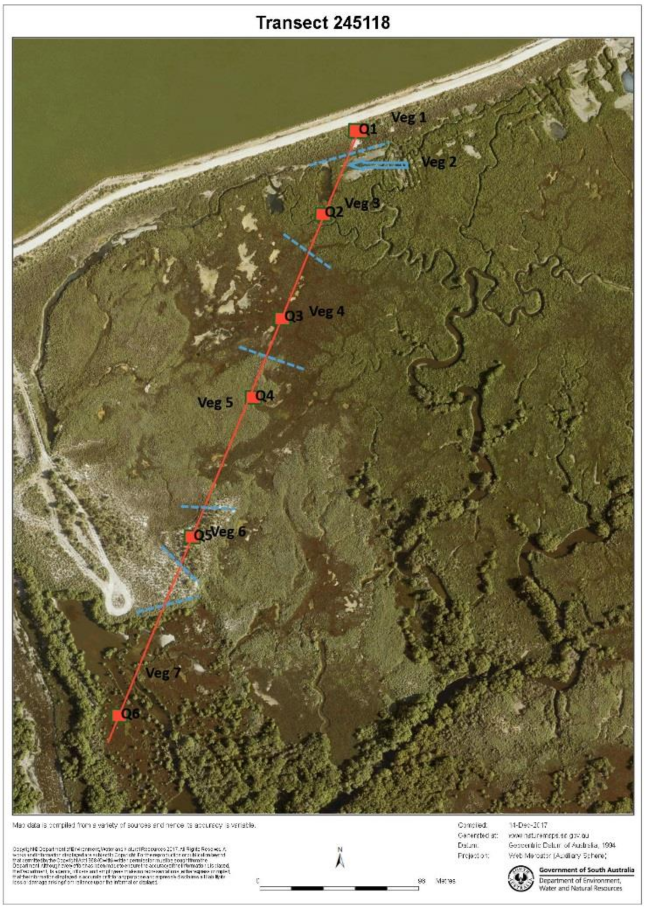

Figure B.1. Aerial view of transect 245118

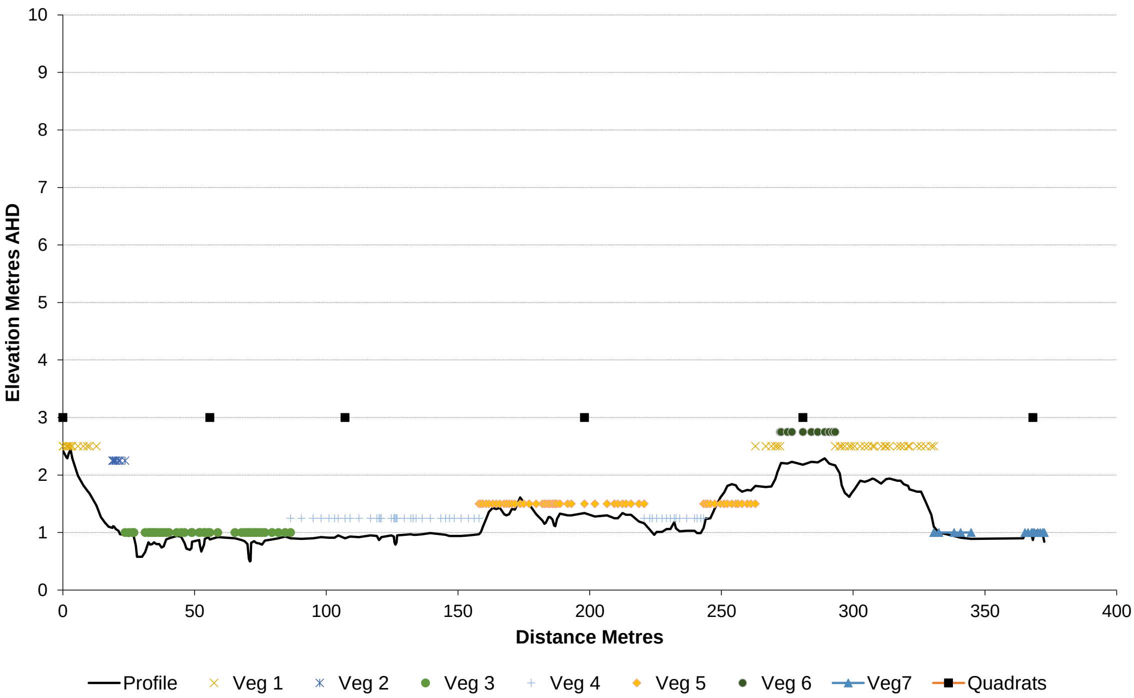

Figure B.2. Profile of elevation and vegetation types along transect 245118 in the reference area.

Figure B.3. Aerial views of transects 245116 and 245117 in the adjacent reference area.

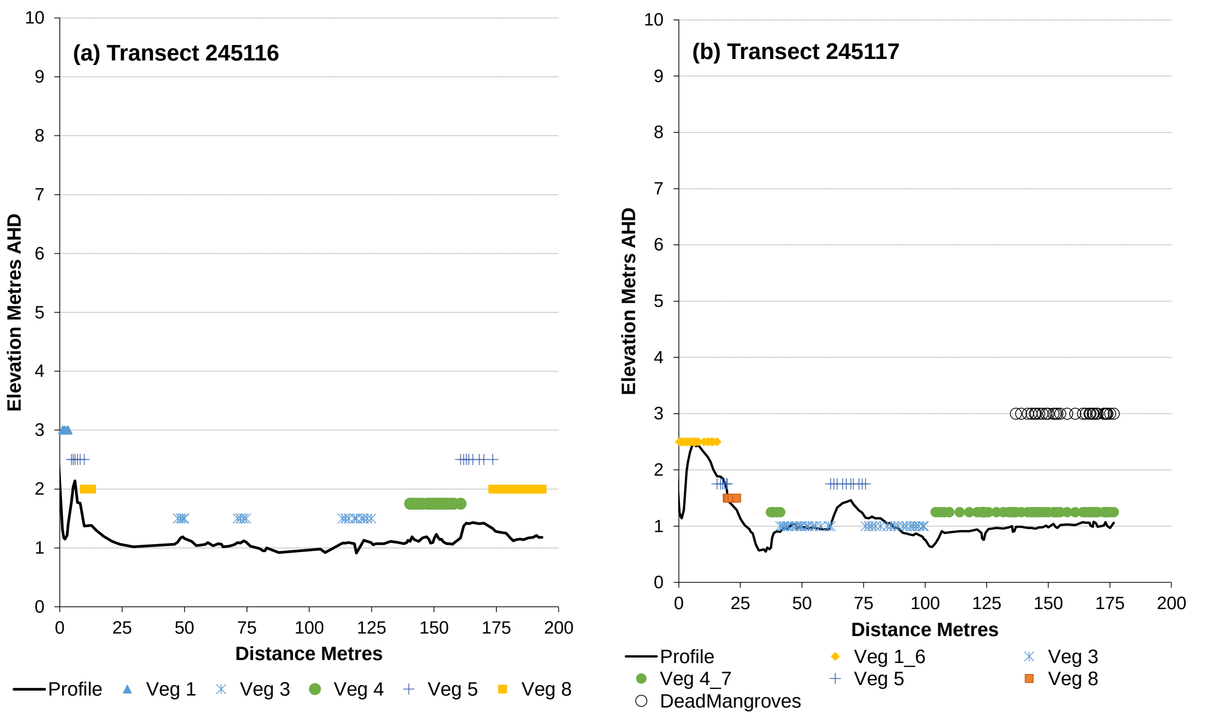

Figure B.4. Profile of elevation and vegetation types along 245116 and 245117 in the adjacent reference area.

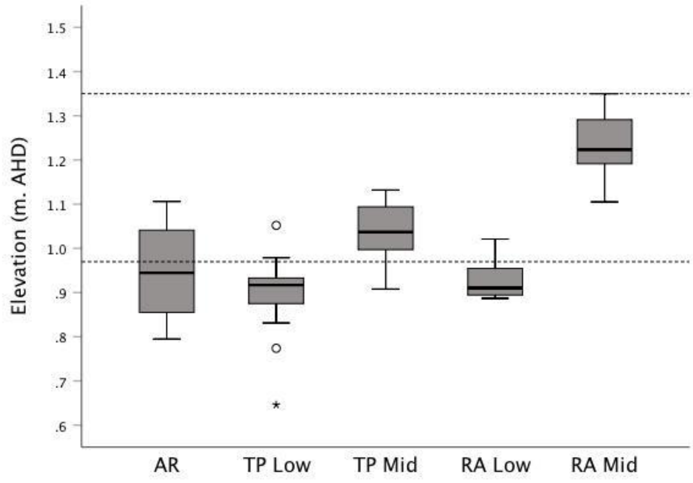

Figure B.5. Box plots of elevation (m AHD) at sample locations for seed trap and seed bank sampling, in the adjacent reference area (AR), the trial pond (TP), and the reference area (RA), for the low and mid elevations. The horizontal lines indicate the upper boundaries for the 'Mangrove-low marsh' (0.97 m AHD) and the 'Tidal saltmarsh' (1.35 m AHD) strata.

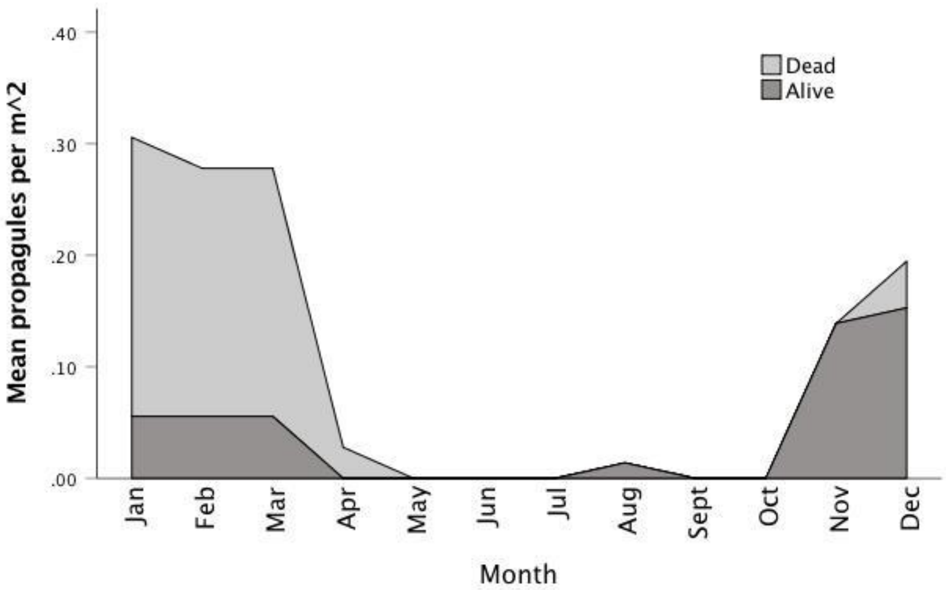

Figure B.6. Phenology of mangrove (*Avicennia marina*) propagules over the course of a year (2018), based on quadrats placed at the study sites.

Table B.1: Mean number of seeds per m^2 within soil seed bank samples (SSB) and seed trap samples (ST), from three sites; Trial pond (TP), Adjacent reference area (AR), and Reference area (RA). Samples were taken from low (MLM, Mangrove-low marsh) and mid (TSM, Tidal saltmarsh) elevation strata. Plant species are separated into core saltmarsh plants and non-saltmarsh plants. SSB samples were taken in February, May, August and November 2018 and seed densities averaged over the sample months. Seed trap samples were taken monthly and seed densities summed over the 12 months.

	POND				ADJACENT		REFERENCE			
	SSB		ST		SSB	ST	SSB		ST	
SPECIES	LOW	MID	LOW	MID			LOW	MID	LOW	MID
Sarcocornia quinqueflora†	9.92	127.31	70.99	1371.6	2026.29	3551.85	745.91	184.15	376.39	516.67
Suaeda australis†	5.79	34.72	32.1	190.74	776.29	1351.85	1.86	5.58	-	1.39
Maireana oppositifolia†	-	-	-	-	-	18.52	3.72	40.92	9.72	191.67
Threlkeldia diffusa†	-	5.79	6.79	209.26	-	-	-	-	-	-
Tecticornia arbuscula†	-	0.83	-	1.23	12.4	22.22	14.88	9.3	4.17	23.61
Frankenia pauciflora†	-	-	-	-	-	-	-	44.64	1.39	1.39
Limonium companyonis†*	-	-	0.62	-	-	-	1.86	11.16	1.39	4.17
Tecticornia pruinosa	-	-	-	-	7.44	-	-	-	-	-
Wilsonia humilis†	-	-	-	-	-	-	-	3.72	-	2.78
Tecticornia halocnemoides	-	0.83	-	-	-	1.85	1.86	1.86	-	-
Mesembryanthemum nodiflorum	-	-	-	-	4.96	-	-	-	-	-
Hemichroa pentandra†	-	-	-	-	-	-	-	1.86	-	1.39
Spergularia media	0.83	-	0.62	-	-	-	-	-	-	-
Saltmarsh abundance	17	169	111	1773	2827	4946	770	303	393	743
Non-saltmarsh abundance	7	17	12	10	35	24	4	7	10	4
Total abundance	24	186	123	1783	2862	4970	774	311	403	747

† Species that were also recorded in the ground cover surveys. Note, the table does not include *Avicennia marina*, which was also recorded in ground cover surveys. * introduced species

Appendix C – Carbon offset registration

C.1 Decision by the Clean Energy Regulator

Decision and reasoning from the Clean Energy Regulator on the tidal trial registration under the '*Human-Induced Regeneration of a Permanent Even-Aged Native Forest*' methodology from August 2017:

Not being able to model Mangroves in FullCAM

Section 27(4)(c) of the *Carbon Credits (Carbon Farming Initiative) Act 2011* (CFI Act) states that the Regulator must not declare that the offsets project is an eligible offsets project unless the Regulator is satisfied that the project meets such requirements as are set out in the methodology determination.

Section 16(4)(b) of the *Carbon Credits (Carbon Farming Initiative) (Human-Induced Regeneration of a Permanent Even-Aged Native Forest-1.1) Methodology Determination 2013 Compilation No. 2* (the method) states that a Carbon Estimation Area (CEA) must only consist of land for which it is possible, in accordance with the FullCAM guidelines, to model the entire CEA to represent the management activities and disturbance events in the area of land.

Section 2.6 of the FullCAM Guidelines sets out the requirements for populating the "Data Builder" tab for a FullCAM plot file. Step 3 requires that the user selects 'Mixed species environmental planting' for the 'Tree species' input. Mixed species environmental planting is defined in the FullCAM calibration paper as eucalypt trees in temperate regions, acacia trees, shrubs, and underground biomass. There is an option to model mangroves in the Tree species input but this option is not allowed under the guidelines.

It is our view that modelling mangroves under the Mixed species environmental planting is not in accordance with the guidelines. As it is not possible to model the CEAs in accordance with the FullCAM Guidelines, it doesn't meet the requirements of the method and as such the project cannot be registered as an eligible offsets project.

C.2 Human-induced regeneration method (HIR)

Table C.1: Emissions and removals for HIR projects in the project boundary (Australian Government Clean Energy Regulator 2018).

EMISSION SOURCES	CORRESPONDING GHG SOURCES
Emissions from and removals to the above and below ground tree and debris pools	• Increases in carbon stocks relating to tree growth • Reductions in carbon stocks relating to biomass decay • Reductions in carbon stocks relating to disturbance – fire or management events (i.e. thinning)
Emissions from project activities	• Emissions from use of fuel to power vehicles and machinery for planning and site selection • Emissions from use of fuel to power vehicles and machinery for management operations, including thinning of trees and fire control (prescribed and unplanned) • Emissions from use of fuel to power vehicles and machinery for transportation and travel (of people or supplies) between business locations, or for deliveries to the project site

Conditions which must be met for HIR projects to generate Australian Carbon Credit Units (ACCUs) (Department of the Environment and Energy 2018):

- The project must be maintained ‘permanently’, up to 100 years.
- In case of fire and other disturbance event, which leads to decline in the amount of carbon stock, the project should be managed to bring the carbon stock to return to previously reported values.
- It involves establishment of permanent native forests through assisted regeneration from *in situ* seed sources, including rootstock and lignotubers.
- It applies to area where land has been cleared of native vegetation and regrowth has been suppressed for at least 10 years.
- The carbon stored in the forest is calculated using a modelling tool, rather than calculated from field measurements.
- The method do not apply to projects that establish a permanent native forest cover by the manual planting of seed or seedlings.
- Even-aged means tree stems that have regenerated as a result of a change to a land management activity at an identified point in time.

Requirements for HIR projects (Australian Government Clean Energy Regulator 2018):

- The project area has not had forest cover (20% crown cover consisting of trees of at least 2 m in height) over the ten years before project commencement due a suppression mechanism (i.e. grazing, mechanical destruction).
- The area of regeneration must have the potential to attain forest cover.
- The regrowth may only be grazed by livestock if the grazing does not materially impact the carbon stocks.
- The project must establish forest cover through the promotion of natural regrowth of vegetation, and not through direct seeding or tree planting.
- Since the method was varied in 2016, conservation land can be eligible for projects under a limited set of conditions. Projects on conservation land must undertake weed or feral animal control and demonstrate that management goes above and beyond what would occur under standard practice.
- The regrowth must not be harvested except for in very limited circumstances such as hazard reduction.
- You cannot establish projects on land that has been cleared unlawfully.

Appendix D – Ecosystem service values, social and cultural values study figures

Table D.1: Some local values of ecosystem services associated with coastal wetlands from Australia in the global database of 1310 data points.

BIOME	ECOSYSTEM	ECOSYSTEM SERVICES	SUBSERVICE	COUNTRY	REGION	YEAR	METHOD	QUERY VALUES	SELECT UNIT(S)	YEAR	QUERY FULL REFERENCE
Coastal wetlands	Mangroves	Food	Fish	Australia	Province/ region	1976	Benefit transfer	1975	USD/ha/yr	1984	Hamilton, and Snedaker (1984)
Coastal	Estuaries	TEV	TEV	Australia	Country	2005	Total economic value	41055.63	AUD/ha/yr	2006	Blackwell (2006)
Coastal	Seagrass/ algae beds	TEV	TEV	Australia	Country	2005	Total economic value	34172.27	AUD/ha/yr	2006	Blackwell (2006)
Coastal	Continental Shelf Sea	TEV	TEV	Australia	Country	2005	Total economic value	2895.04	AUD/ha/yr	2006	Blackwell (2006)
Coastal wetlands	Mangroves	TEV	TEV	Australia	Country	2005	Total economic value	17963.64	AUD/ha/yr	2006	Blackwell (2006)
Coastal	Seagrass/ algae beds	Nursery	Nursery service	Australia	Local	2001	Factor income/ production function	133.23	US$/ha/yr	2006	McArthur, & Boland (2006)
Coastal wetlands	Mangroves	Nursery	Nursery service	Australia	Landscape / district	2007	Direct market pricing	5846.52	USD/ha/yr	1990	Morton (1990)

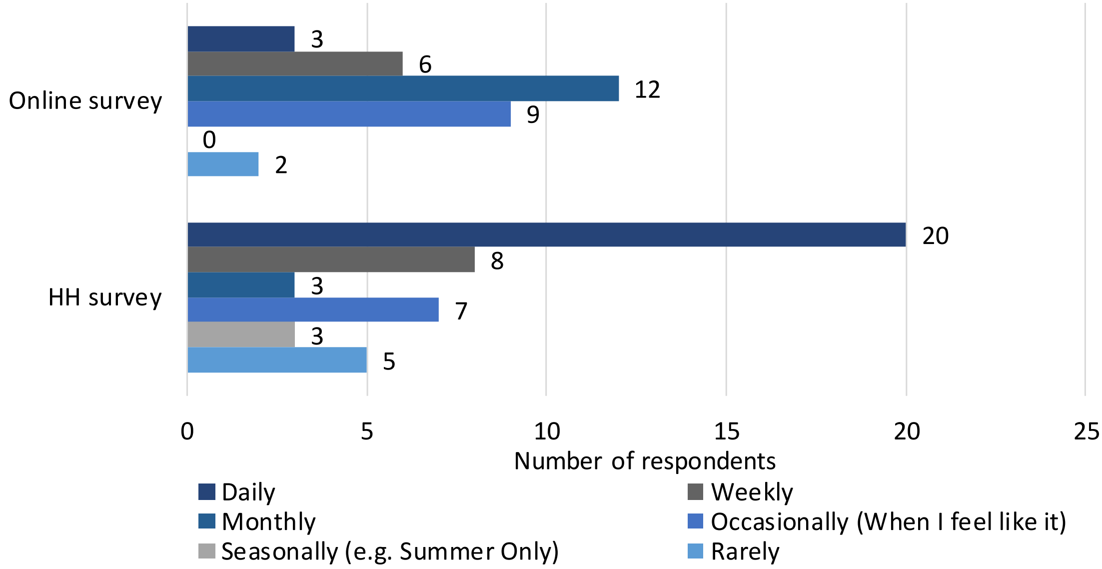

Figure D.1. Visiting frequency by survey type (number of respondents). HH = Household survey.

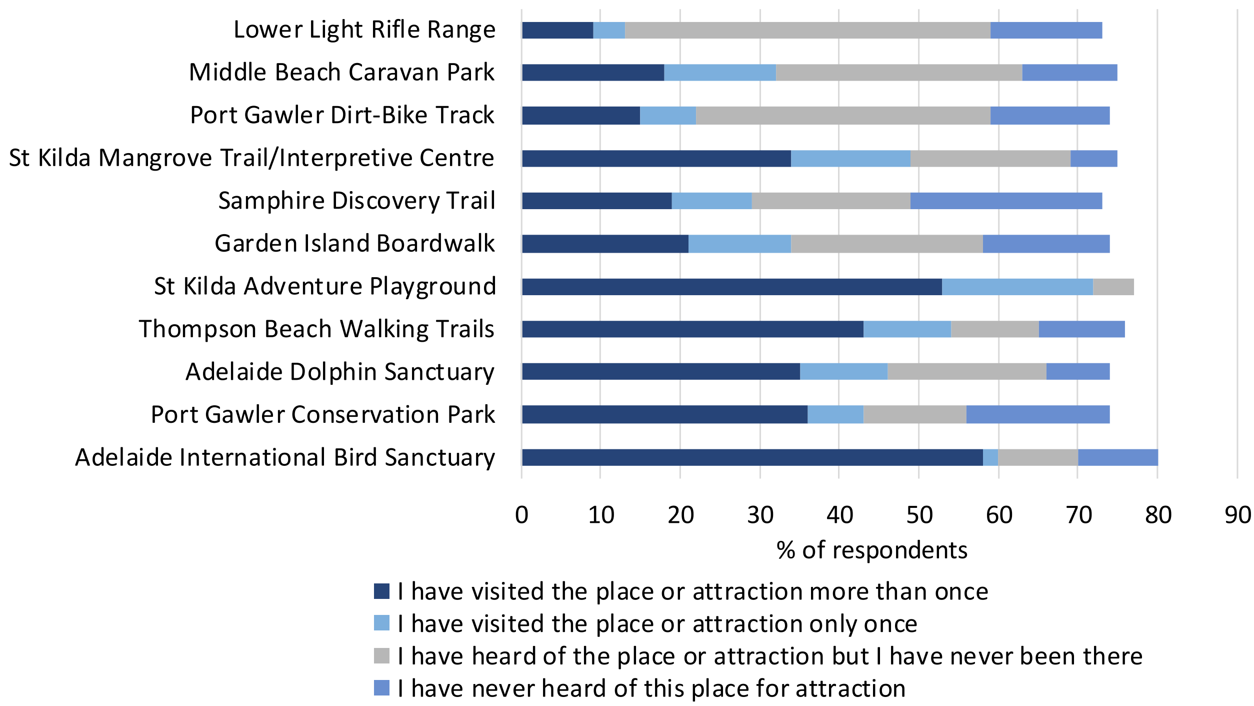

Figure D.2. Sites by visit frequency (% of respondents).

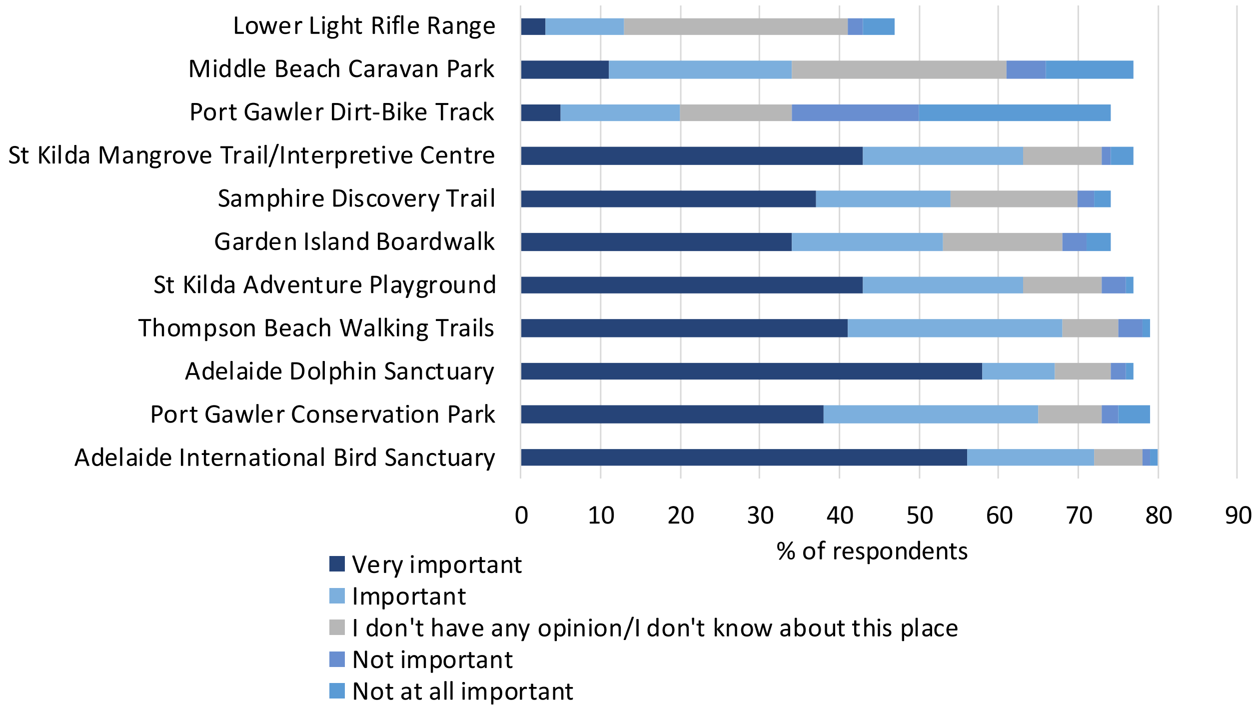

Figure D.3. Sites by importance rating (% of respondents).

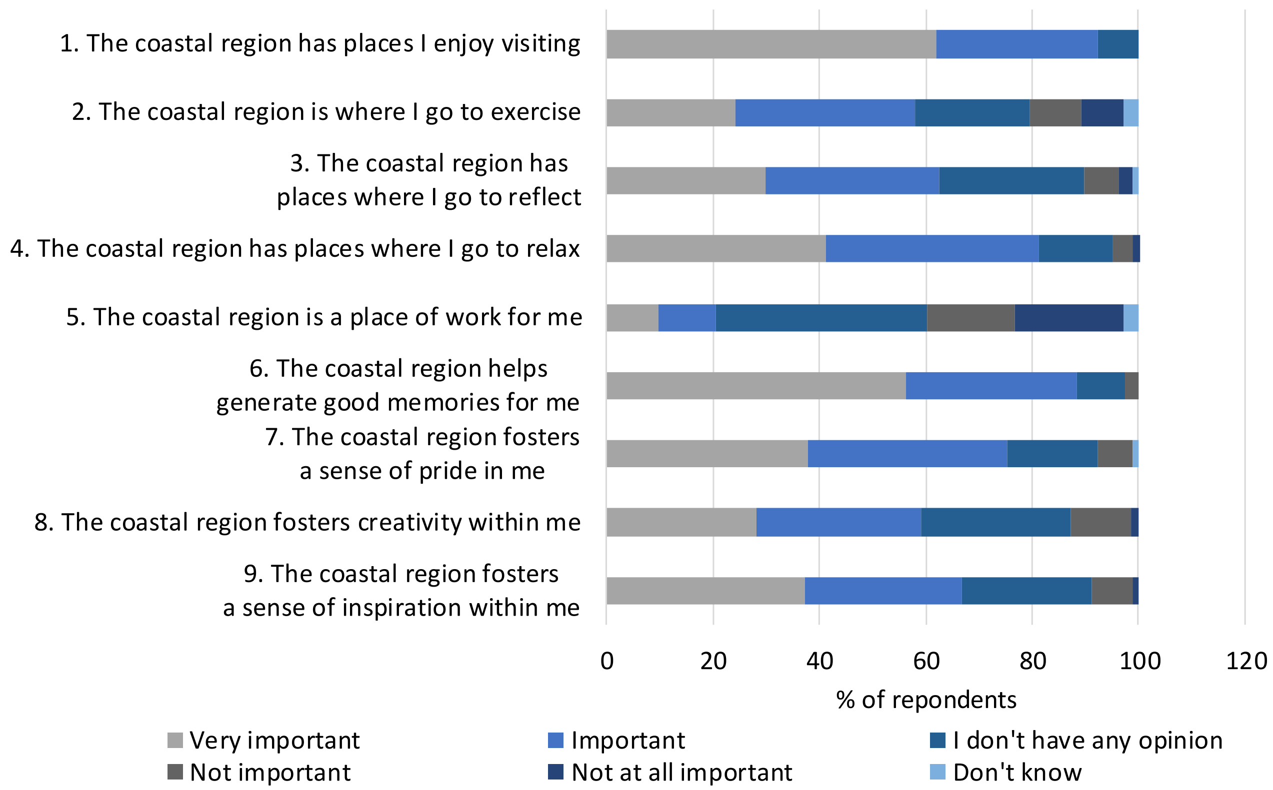

Figure D.4. Personal rating of well-being (% of respondents).

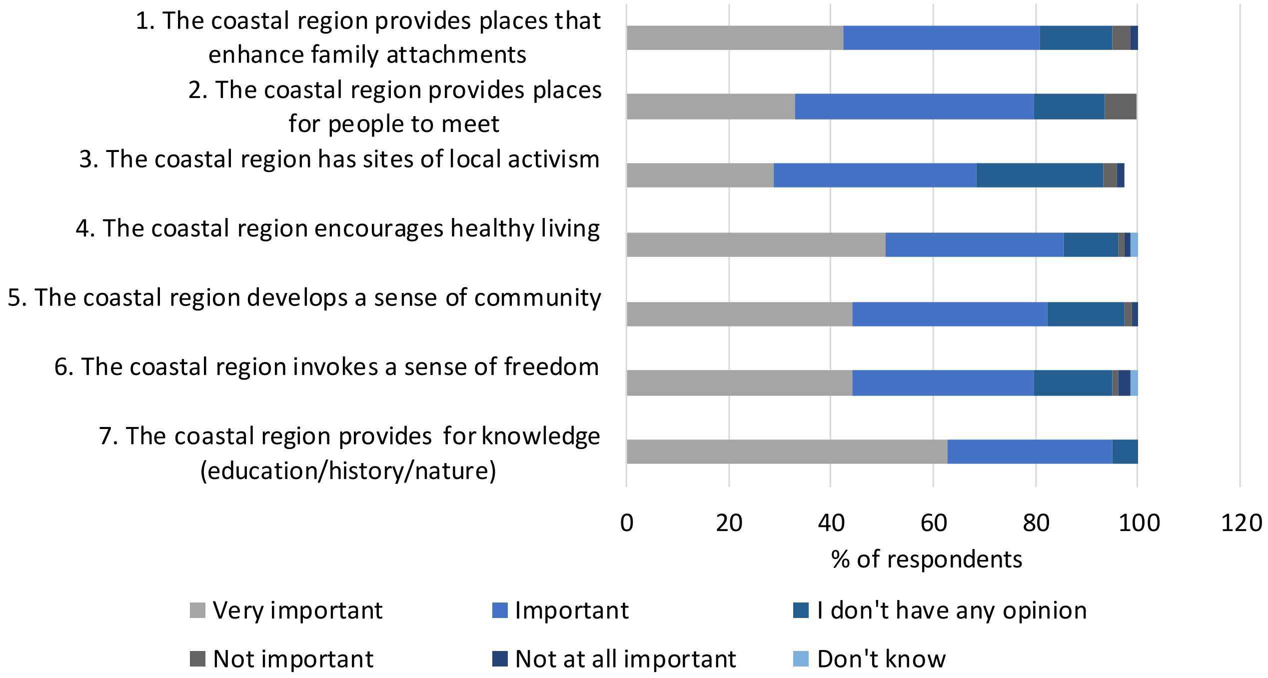

Figure D.5. Rating of cultural value aspects associated with the coast between Torrens Island and Thompson Beach (%).

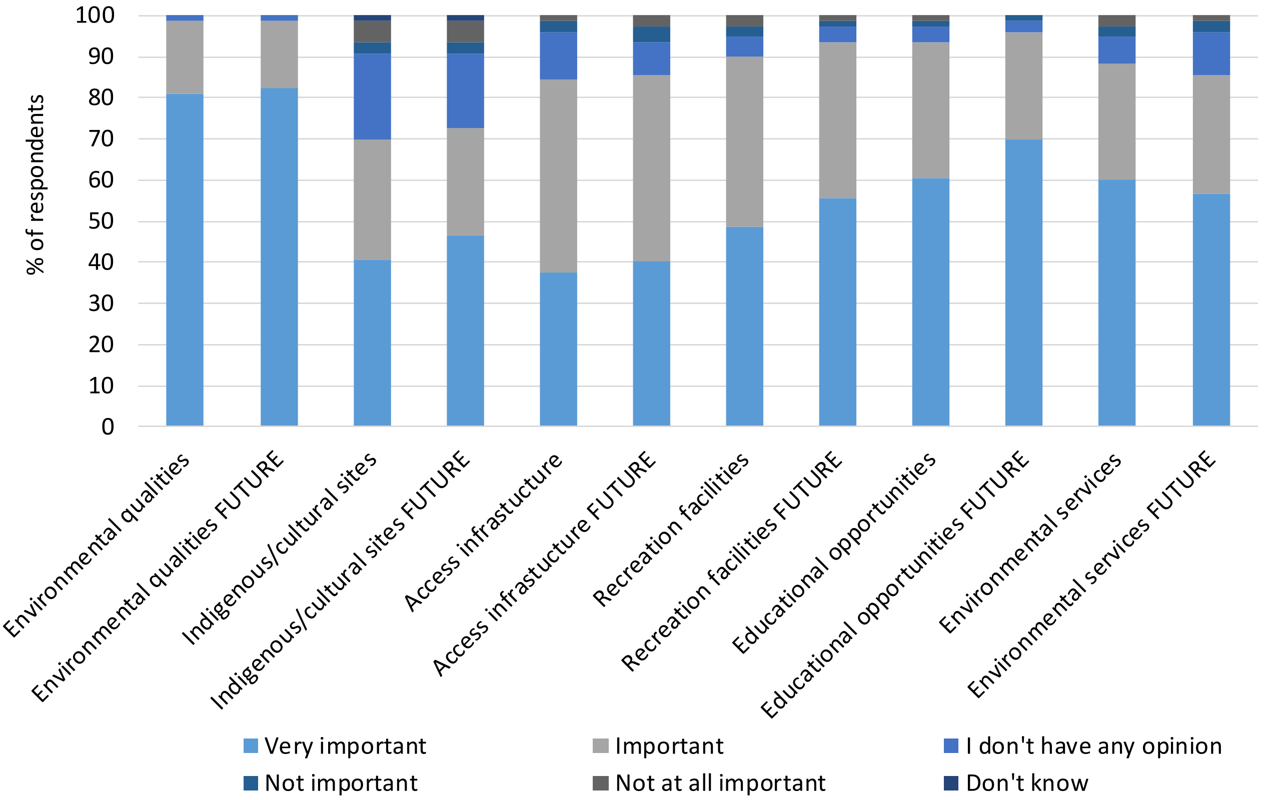

Figure D.6. Importance of assets and qualities within the study region.

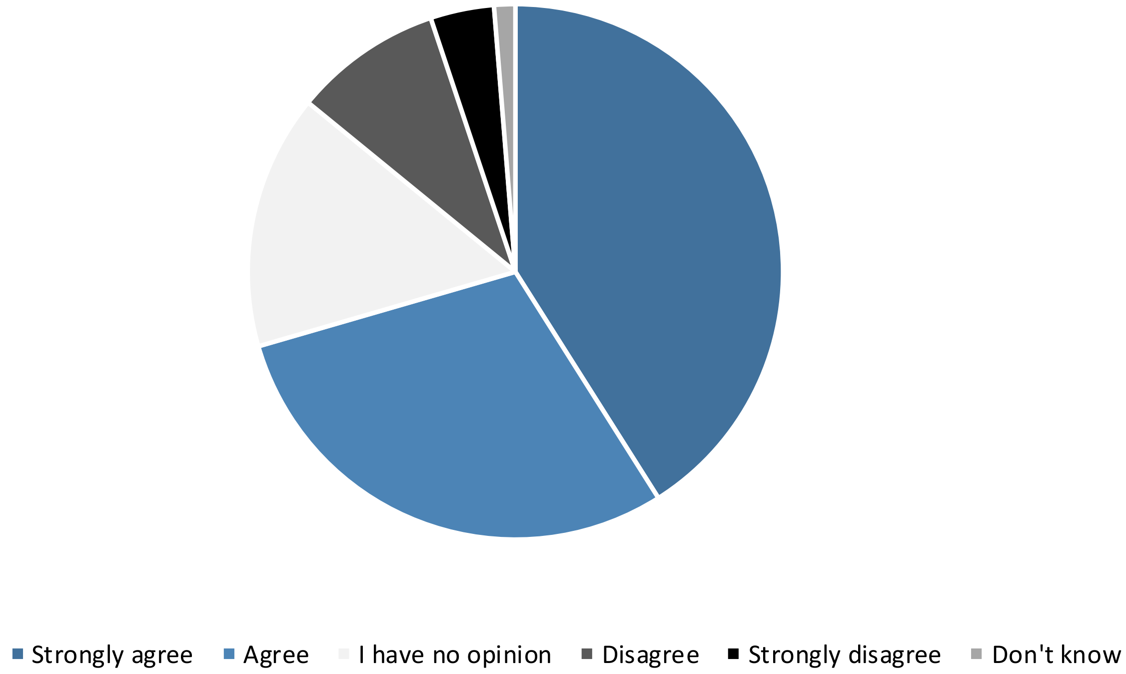

Figure D.7. Concerned that impacts of erosion, flooding and/or storms will affect personal use of the coastal region between Torrens Island and Thompson beach.

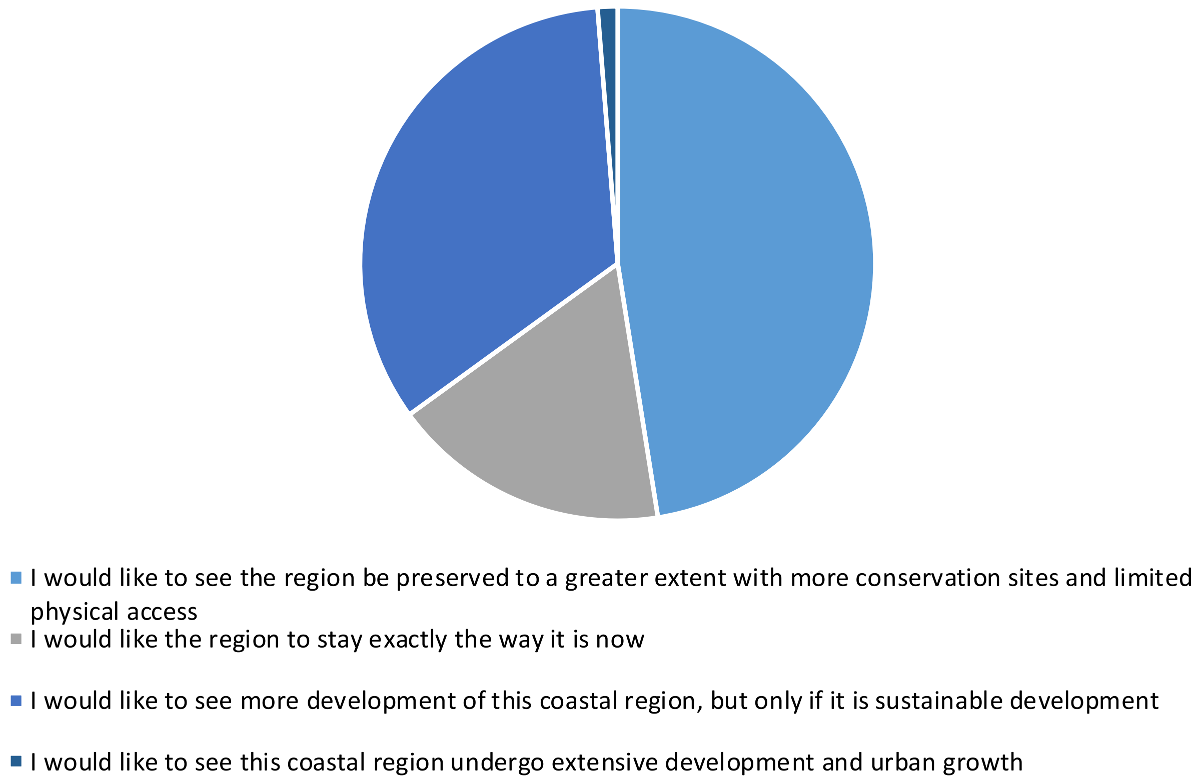

Figure D.8. Preference for the future of the coastal region between Torrens Island and Thompson Beach.

Table D.2: Overview of three scenarios considered and the area for each scenario, split by strata MLM = 'Mangrove-low marsh', TSM = 'Tidal saltmarsh', SSM = 'Supra-tidal saltmarsh', from the lower limit of mangrove (0.44 m AHD) to 2.1 m AHD, i.e. creeks <0.44 m and elevations >2.1 m AHD are not included). The estimated net gains in carbon stocks are given for the carbon pools of soil and biomass, assuming at least 30 years of recolonization time.

			NET GAIN IN CARBON STOCKS			
			t C		t CO_2e	
SCENARIO	STRATA	AREA (ha)	SOIL	BIOMASS	SOIL	BIOMASS
Scenario 1: Trial pond reconnected	MLM	20	373	2,368	1,367	8,684
	TSM	6	26	142	97	520
	SSM	4	17	26	64	96
Scenario 2: Low-lying ponds reconnected	MLM	473	8,903	33,344	32,645	122,260
	TSM	532	2,347	12,632	8,607	46,318
	SSM	205	903	1,364	3,313	5,003
Scenario 3: Higher-lying ponds reconnected	MLM	480	9,029	34,068	33,107	124,915
	TSM	560	2,469	13,288	9,054	48,724
	SSM	931	4,108	6,203	15,061	22,744

The Goyder Institute for Water Research is a partnership between the South Australian Government through the Department for Environment and Water, CSIRO, Flinders University, the University of Adelaide, the University of South Australia, and the International Centre of Excellence in Water Resource Management.